3-1986

Structural and Functional Organization of Macroarthropod Communities in a Mined Versus a Pristine Stream in the Cumberland Mountains of Eastern Tennessee

Joseph R. Schiller

To the Graduate Council:

I am submitting herewith a dissertation written by Joseph R. Schiller entitled "Structural and Functional Organization of Macroarthropod Communities in a Mined Versus a Pristine Stream in the Cumberland Mountains of Eastern Tennessee." I have examined the final electronic copy of this dissertation for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Doctor of Philosophy, with a major in Animal Science.

Gerald R. Vaughan, Major Professor

We have read this dissertation and recommend its acceptance:

David A. Etnier, Susan E. Riechert, Larry Wilson

Accepted for the Council:
Dixie L. Thompson

Vice Provost and Dean of the Graduate School

(Original signatures are on file with official student records.)

To the Graduate Council:

I am submitting herewith a dissertation written by Joseph R. Schiller entitled "Structural and Functional Organization of Macroarthropod Communities in a Mined Versus a Pristine Stream in the Cumberland Mountains of Eastern Tennessee." I have examined the final copy of this dissertation for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Doctor of Philosophy, with a major in Zoology.

Gerald L. Vaughan

G. L. Vaughan, Major Professor

We have read this dissertation
and recommend its acceptance:

Laurence Eisner

Susan E. Riechert

Gary Wilson

Accepted for the Council:

Lawrence Winkel

Vice Provost
and Dean of The Graduate School

STRUCTURAL AND FUNCTIONAL ORGANIZATION OF MACROARTHROPOD COMMUNITIES IN A MINED VERSUS A PRISTINE STREAM IN THE CUMBERLAND MOUNTAINS OF EASTERN TENNESSEE

A Dissertation

Presented for the

Doctor of Philosophy

Degree

The University of Tennessee, Knoxville

Joseph R. Schiller

March 1986

ACKNOWLEDGEMENTS

In the course of dissertation research I have been assisted by friends, family, and faculty in ways too numerous to ever fully acknowledge here. I would like to thank the members of my graduate committee Drs. Gerald Vaughan, David Etnier, Susan Riechert, and Larry Wilson for their support and guidance. Dr. Dewey Bunting generously provided lab facilities and equipment which were essential to this study. Lurae Minter, Patty McCall, Steve Myhr, Ward Wampler, and Pete Kalla deserve special thanks for assisting me in long (and often wet) days of field work. Pete has also been most helpful with editing. Wendell Pennington has graciously identified the chironomid fauna of Crabapple and Bruce Creeks. My parents and inlaws have been very encouraging and supportive over the years, and I can never thank them enough. Most of all I want to thank my wife, Sally Schiller, for no one person deserves as much credit as her in helping me to complete this dissertation.

This research was partially supported by USDOE contract # EY-76-S-05-4946.

ABSTRACT

The effects of surface mining coal on the structural and functional organization of macroarthropod communities as a result of changes in the physical structure of small streams was investigated. Two first order streams in the Cumberland Mountains of eastern Tennessee, a pristine reference stream and a stream mined approximately 15 years earlier, were compared for particulate organic matter (POM) dynamics, structural and functional group composition, and productivity of their macroarthropod communities. POM dynamics of the two streams were compared by measuring their input, standing stock, and processing in each stream. Macroinvertebrates were sampled with standard Surber samplers. The community structure of the two streams was compared in terms of taxonomic composition, abundance, species richness, and Shannon diversity. Functional group composition of the macroarthropod communities were compared using the functional group classification of Merritt and Cummins (1984). Productivity of the dominant taxa was calculated by the size-frequency method. The total productivity and productivity within functional feeding groups were then compared.

Input and standing stock of POM in the two streams was similar. Faster rates of POM processing in the mined stream resulted from greater macroarthropod shredder densities. Differences in POM processing, macroarthropod functional group composition, and productivity within functional groups between the two streams suggest that their macroarthropod communities are functionally different. The lower abundance, species richness, and diversity were typical of the structural differences reported for other macroarthropod communities in mined streams of

this region. Overall productivity of the surface mined stream was similar to the reference stream, although it was distributed differently between functional groups.

The results indicated that analysis of the functional group organization of mined streams reveals important differences in their ecology which could be underestimated or overlooked by conventional indices of community structure. Taxonomic surveys and diversity indices provide useful tools for monitoring and detection of environmental degradation. However, additional understanding of the causal relationships between environmental disturbance and biological change can be obtained from a functional approach which can identify its effects on specific biological processes in the stream community.

TABLE OF CONTENTS

LIST OF TABLES

TABLE PAGE

LIST OF FIGURES

CHAPTER 1

INTRODUCTION

Background and History

The impetus for this study originates out of concern for the effects of surface mining coal on the ecology of small streams in the southern Appalachians. The Appalachian coal field (eastern coal province) is one of four major coal basins in the eastern U.S. The ecological consequences of surface mining coal vary in different areas of the country because of regional differences in topography, geology, and climate. Adverse ecological effects due to surface mining coal may reach their greatest dimensions in the Appalachian coal fields because of the unfavorable combination of steep terrain, plentiful rainfall, and relatively thin but numerous coal seams. These conditions make surface mining in this region technically difficult and reclamation more so. Problems associated with mining and reclamation are reflected in the fact that 68% of the abandoned mine lands of the eastern U.S. are located in this coal field (U.S. Soil Conservation Service 1979).

While adverse ecological consequences are certainly worse when mining proceeds unregulated and without adequate reclamation, it is not established that even conscientious reclamation, as currently required, protects against severe ecological degradation of streams in this region. Minear et al. (1980) studied recontoured stripmine spoil slopes which had uniform, heavy stands of grass growing on them and conformed to state and federal reclamation practice regulations. They measured sediment yields of approximately 50 ton/acre/year. Every storm event

monitored resulted in suspended sediment concentrations greater than the Tennessee's maximum allowable 100 mg/L (range: 264 - 103,100 mg/L). It was observed by Matter and Ney (1981) that streams draining abandoned versus reclaimed mines in southwest Virginia differed little in water quality and benthic community structure when acid mine drainage was not involved. They concluded that "terrestrial reclamation does not assure lotic restoration". Their findings do not mean that surface mined watersheds fail to recover but rather that current reclamation requirements do not mitigate the adverse environmental effects of surface mining on biological communities. Contributing to this problem is a historical lack of enforcement of existing regulations which considerably compromise any possible beneficial effect of mine reclamation. In addition, a considerable amount of "wildcat" mining occurs completely outside of the regulatory process.

A large body of water quality data has been compiled by various state and federal agencies to serve as baseline data for water quality monitoring and impact assessment of mining and other stream polluting activities in this region. In fact there is a USGS gauging station in the present study area that is part of a network of water quality stations covering the entire eastern coal province (essentially all of Appalachia). The limited data released thus far for Crabapple Creek, the undisturbed reference stream of this study, are comparable to the data reported here (U. S. Geological Survey 1983).

Dyer (1982) surveyed 69 small streams in Tennessee counties where coal has been mined. The streams were categorized as unmined, mined before 1972, and mined after 1972. This report contains data on a wide

range of physical and chemical parameters collected approximately monthly over a two-year period. While it contains little background and site description information other than topological site maps, it does provide basic physical and water chemistry data for streams affected by a wide range of mining conditions. The data reflect the general water quality patterns observed in other studies, i.e. that suspended solids, turbidity, alkalinity, and hardness are generally higher in mined streams, and that the concentrations of sulfate, iron, manganese, and some other chemicals are increased in mined streams, but that most substances do not reach toxic levels. Dyer and Curtis (1977) provided comparable data for small streams in eastern Kentucky.

Leist et al. (1982) provided data for 38 hydrologic reporting stations in the upper Cumberland River drainage, Area 15 of the eastern coal province. This report, which was part of a series covering the entire eastern coal province, provided a brief summary of virtually all topics relevant to documenting the chemical and hydrological effects on streams of surface mining coal in this region. They found that specific conductance provided a reliable estimate of suspended solids and that the highest values of these parameters were found in areas with the most mining activity. The pH of most streams was near neutral. Acid mine drainage, when it occurred, was usually neutralized by carbonaceous rock close to its source, and the pH of some streams was increased by mining activity. Sulfate, iron, and manganese concentrations were generally low in the relatively unmined Rockcastle River basin but much higher in the heavily mined headwaters of the Cumberland River basin in Harlan and Bell counties, Kentucky. Suspended sediment yields for the heavily

mined Yellow Creek basin were from five to ten times higher than in the Rockcastle River basin. The transport of most of the annual suspended sediment load in mined streams tended to occur during just a few storm events during the year, an observation that was also made by Minear et al. (1980). This is due to the greater runoff and higher peak flows occurring in mined watersheds. For example, average runoff from the Rockcastle River basin was 40% of precipitation but average runoff in the Cumberland basin was 50% of precipitation. Curtis (1972a) also noted increased runoff from surface mined streams in southeastern Kentucky.

These studies indicate that the physical effects of surface mining on aquatic environments in the Appalachian coal fields are relatively predictable and can be generalized as a combination of altered stream hydrology and increased sedimentation. The concentrations of some elements and compounds, especially sulfate, manganese, and iron, are increased and provide good indicators of mine disturbance, but rarely reach toxic levels. Acid mine drainage and its associated mobilization of heavy metals occurred mostly in association with deep auger mining in eastern Tennessee. This auger mining often occurred in conjunction with or subsequent to surface mining (Boccardy and Spaulding 1968, Curtis 1977). However, auger mining is no longer practiced in Tennessee. In the absence of acid mine drainage stream water is often alkaline and its pH may increase (Curtis 1972a, Minear and Tschantz 1976, Dyer and Curtis 1977, Leist et al. 1982, Vaughan et al. 1982). Despite this apparent lack of toxic agents, the combined effects of altered hydrology, increased sedimentation, and increased suspended solids resulted in drastic changes in stream ecosystems.

The biological changes resulting from surface mining coal in the southern Cumberland Mountains have been well characterized. Drastic reductions in the density and species diversity of the diatom, fish, and macrobenthic communities follow the onset of mining (Carter 1964, Hensely 1970, Branson and Batch 1972, Talak 1977, Tolbert 1978, Matter and Ney 1978, Vaughan et al. 1978, 1982). Note that all references to species diversity in this dissertation imply diversity indices derived from information theory, specifically, the Shannon-Weaver Index (Pielou 1966). In the absence of acid mine drainage, stream biological communities gradually recover density and species diversity approaching pre-mining values (Talak 1977, Vaughan et al. 1978, 1982).

This dissertation represents a continuation of previous research under the direction of Dr. Gerald Vaughan which has resulted in the production of two M.S. Theses (Talak 1977, Williams 1981), a Ph.D. dissertation (Tolbert 1978), and various publications (Minear et al. 1976, 1977, 1978, 1981, 1982, Stair and Tolbert 1979, Vaughan 1979, Vaughan et al. 1978, 1982). Over the course of these several studies, water quality, fish, diatoms, and aquatic macroinvertebrates were sampled in more than 20 streams in eastern Tennessee over a period of several years. A comprehensive summary of this research is available in Vaughan et al. (1982). Sampling of diatoms was discontinued early in this effort because diatom numbers were reduced to such an extent in mined streams that adequate sample sizes were difficult to achieve (Vaughan 1979). The diatoms, therefore, represent a significant exception to the general recovery of stream biological communities of surface mined streams as exemplified by the fish and macroinvertebrates.

The chronology of stream recovery was documented by Talak (1977) and Vaughan et al. (1978) by sampling the fish, aquatic insects, and water quality of 23 streams that varied in the time since mining ceased, including three unmined reference streams. They found that species diversity, number of taxa, and total density of fish and aquatic insects were lowest in streams mined a few years previously but increased after that, and streams mined 20 or more years previously were similar to reference streams.

Most species of aquatic macroarthropods were less abundant in mined streams, but some taxonomic groups were disproportionately reduced or eliminated altogether. Consequently the taxonomic composition of mined streams usually differs from unmined streams both in the relative abundance and presence of taxa (Talak 1977, Tolbert 1978, Williams 1981, Vaughan et al. 1982). Ephemeroptera were usually the numerically dominant aquatic insect order in the small unmined streams of this region. Although they usually remain numerically dominant, their numbers were disproportionately lower in mined streams relative to other orders of aquatic insects, especially plecopterans. Coleopteran abundance was also disproportionately lower in surface mined streams. Matter and Ney (1981) found differences in the taxonomic composition of mined streams in southwestern Virginia, including disproportionately fewer coleopterans. Talak (1977) found that differences in taxonomic composition of aquatic insects between unmined and mined streams decreased with time after mining.

Even the macroinvertebrate communities of some streams may not completely recover the previous taxonomic composition and/or productiv-

ity that existed before mining. For example, Williams (1981), in a subsequent analysis that included three of the 23 streams studied by Talak, found comparable species diversity values, but lower densities and different taxonomic compositions in two streams mined 25 years prior to sampling compared to two unmined reference streams. Her results suggest that surface mining may leave a long term signature on stream ecosystems consisting of reduced densities and/or altered taxonomic composition. A possible explanation for such long term impact is that surface mining coal often affects streams from their uppermost reaches, eliminating any refugia from which colonizers can drift. The macroarthropod communities of streams suffering equally traumatic stress in other pollution situations, but with unaffected upstream reaches, have quickly recovered upon its removal. For example, Herricks and Cairns (1974) acidified a small section of stream in Virginia. They noted density and diversity reductions comparable to those observed in the macroarthropod communities of surface mined streams in eastern Tennessee, but they recovered rapidly (within weeks) after the acidification treatments were stopped. Macroarthropods have also recovered rapidly following the reduction in discharges of coal ash effluent from coal fired power plant settling ponds into a stream in Virginia (Cherry et al. 1979, Specht et al. 1984).

Much of the past research on the ecology of freshwater, including surface mined streams, has concentrated on measuring structural changes in biological communities. The complex of individuals belonging to the different species in the ecosystem is referred to as community structure (Wilhm and Dorris 1968). Structure of biological communities is often

summarized as species diversity which reflects both the number of species (species richness) and the relative number of individuals in each species (evenness), although structure may also be expressed in the form of a more cumbersome taxonomic census, or simply as species richness. The extensive use of diversity indices for environmental monitoring and impact assessment has come under criticism from some investigators who question its usefulness relative to more straightforward statistical methods (Green 1979). Another problem with the use of diversity indices is in comparing them between different studies. Different ecosystems and communities within ecosystems can have different characteristic diversity. One of the reasons diversity indices between studies are difficult to compare is inconsistent taxonomic classification. Perhaps the biggest drawback to the use of diversity indices is loss of information resulting from the compression of all the species abundances into a single index.

Proponents of diversity indices maintain that they reliably reflect changes in the structure of aquatic biological communities resulting from environmental degradation and that they offer a concise medium for communicating experimental and environmental monitoring results (Wilhm and Dorris 1968, Weber 1973). Measuring environmental degradation as a biological parameter has the potential advantage of detecting transient or unknown environmental pollutants which could be missed by analyzing water samples for physical and chemical parameters. This latter point is largely irrelevant to the stripmining situation because its effects are chronic, not intermittent, and easily detectable from water sample analysis. However, the degree to which the observed water quality

parameters affect the streams biological communities is well described by diversity. Talak (1977) and Vaughan et al. (1978) have shown that diversity of biological communities in eastern Tennessee streams reflects the impact of surface mining and the degree to which it moderates over time after mining ends. Earlier work by Wilhm and Dorris (1966) related differences in diversity along the length of Skeleton Creek in Oklahoma to the effects of organic enrichment from domestic sewage and oil refinery effluent. Diversity was reduced in upstream areas close to the discharges and increased downstream as their effects were dissipated. They also provided evidence for the value of diversity indices in environmental monitoring from several other pollution studies (Wilhm and Dorris 1968). Herricks and Cairns (1974) noted decreased diversity of stream macroinvertebrate communities in response to acid mine drainage in Indian Creek in Pennsylvania. Low diversity was obtained from areas receiving acid mine drainage but increased downstream from the confluence of unpolluted tributaries which neutralized or diluted the acidity. The results of these and other studies support the use of diversity as a valuable tool for environmental monitoring and impact assessment.

More recently the need to consider the effects of environmental degradation on the physical structure of streams and the subsequent changes in the functional organization of stream communities has been proposed (Cummins 1974). Note the physical structure of streams and the structure of biological communities are separate concepts, although one may affect the other. The physical structure of streams includes properties such as size, morphology, substrate, and hydrology. These param-

eters interact with or are determined by watershed characteristics. The functional properties of biological communities describe how the community is organized to process energy and nutrients. One of the advantages of a functional approach to studying stream ecology is that members of the stream community can be categorized by their functional role in the community, at least partially avoiding some of the taxonomic problems involved in diversity analysis (Cummins 1974).

There are several ecosystem processes in streams in which different groups of the stream biota perform distinct functional roles. Obviously, as in terrestrial ecosystems, there is the process of primary productivity, i.e., photosynthesis, which is performed by plants and algae. However, the carbon fixed in the stream by photosynthesis, autochthonous carbon, is usually not the most significant source of carbon supporting the consumer organisms of stream ecosystems. Usually most of the energy base of lotic ecosystems is allochthonous carbon, carbon imported from outside the stream. This allochthonous carbon is mainly in the form of sensescent plant parts: leaves, wood and fruits; insect frass and parts; and dissolved organic matter (DOM). In some studies any carbon imported from upstream is also considered allochthonous to distinguish it from the carbon fixed in the specific stream section under study. The particulate fraction of this allochtonous carbon is often referred to collectively as simply detritus or particulate organic matter (POM). Detritus larger than 1 mm is defined as coarse particulate organic matter (CPOM); and detritus smaller than 1 mm is defined as fine particulate organic matter (FPOM). This rather arbitrary size boundary corresponds approximately to the difference between organic

matter which must be chewed up and broken down versus organic matter which can be collected and ingested.

Because of the importance of detritus in most lotic ecosystems, several consumer functional groups involved in its utilization have been distinguished (Cummins 1974, Boling et al. 1975a, 1975b). Therefore, secondary productivity in lotic ecosystems results from the consumption of stream plants and algae, grazing, by a "grazer" functional group, and from the consumption of detritus by additional functional groups. CPOM is utilized by a succession of functional groups in the detritus pathway. Aquatic hypomycetes are the most important stream microbes in the actual digestion of CPOM and wood but bacteria and other microorganisms also colonize it (Fairchild et al. 1983). The establishment of this microbial-detrital complex has been termed "conditioning" because it renders detritus more palatable and nutritious to macroinvertebrate consumers (Kaushik and Hynes 1971, Cummins 1974, Petersen and Cummins 1974, Barlocher and Kendrick 1975, Kostalos and Seymour 1976, Short and Maslin 1977). Macroinvertebrates which chew up and ingest CPOM are designated shredders, for the shredding function they effect in their feeding. Macroinvertebrates which utilize FPOM, much of it originating as uningested or undigested particles created by shredders, are designated collector-gatherers if they search the streambed, or collector-filterers if they strain the flowing water to obtain their food. The unidirectional movement of materials resulting from the downstream flow of water imposes a unique organization on lotic communities and makes possible the filter-feeding functional group for which there is no exact terrestrial analogue.

Cairns et al. (1973) considered the merits of community structure versus community function in environmental monitoring of aquatic ecosystems. At that time they considered the methodology and techniques for the study of community function to be insufficiently developed and that the structural integrity of aquatic ecosystems was a reflection of its functional condition, rendering its specific study redundant. In this dissertation structural and functional properties of stream biological communities are not assumed, a priori, to be redundant. It was assumed instead that each approach is capable of yeilding separate insights into the effects of surface mining coal on stream biological communities. Since watershed disturbance from surface mining coal results in alteration of the physical structure of streams which may in turn result in alterations of the functional organization of stream communities, a functional approach to the study of mining effects on lotic ecosystems seems justified.

Changes in the physical structure of streams due to siltation and hydrologic factors are considered the probable agents of biological change in most surface mined streams in eastern Tennessee (Tolbert 1978, Vaughan et al. 1982), but exact causal relationships remain unexplained. In a few areas in eastern Tennessee and southeastern Kentucky acid mine drainage occurs as a result of surface mining, but it is the exception rather than the rule (Minear et al. 1976, Dyer and Curtis 1977, Leist et al. 1982).

Siltation alters the physical structure of the stream by filling in pools and interstitial spaces and creating an unstable substrate. Hydrologic factors, such as larger and more frequent floods, contribute

to the destabilization of the stream bottom. Stair and Tolbert (1979) reported stream flow velocities over two times faster in small surface mined streams compared to similar unmined reference streams during storms. This observation combined with the results of a velocity tolerance study of several species of aquatic insects inhabiting these streams led them to conclude that increased stream flows during storm events could be an important factor in the reduction of macrobenthos densities of mined streams.

The increased stream flows of mined steams are capable of moving large rocks considerable distances (Minear et al. 1977, Stair and Tolbert 1979). It seems reasonable to expect that such stream power would also remove debris dams, large aggregations of leaves, wood, and other organic matter, from the stream channel. Debris dams are an important structural feature of small forested streams. Debris dams contained 75% of the standing stock of CPOM in the first order New Hampshire streams studied by Bilby and Likens (1980). Artificial removal of debris dams from stretches of a small stream in the Hubbard Brook watershed in New Hampshire resulted in a five-fold increase in transport of CPOM and FPOM through the stream (Bilby 1981). The more frequent and severe floods of surface mined streams might therefore reduce their retentiveness of organic matter thereby reducing their overall productivity.

These structural effects of surface mining on streams could affect the functioning of stream communities in several ways. Siltation can in rare cases reach such drastic proportions that many organisms are literally smothered. Branson and Batch (1972) found salamanders entombed

under rocks covered by several centimeters of silt in east-central Kentucky. More often a thin layer of silt accumulates on the stream bottom during the declining phase of the stream hydrograph. This thin layer of silt might interfere with the exchange of nutrients, metabolites, and respiratory gases between flowing stream water and benthic microbial populations because of increased diffusion distances. Reice (1974) observed that leaves were processed at slower rates on silt substrates in streams. Perhaps silt interferes with microbial digestion and conditioning thereby resulting in slower overall processing. Stream microbial communities also utilize DOM, converting it into POM by their assimilation of it. This organic matter is then retained in the stream and remains available to macroinvertebrates (Fisher and Likens 1973, Lush and Hynes 1973, McDowell and Fisher 1976, Wallis 1980). DOM originates from leaf leachates, from organic matter washed out of the forest canopy, and perhaps most importantly from groundwater (Hynes 1983). The basic ecology of the hyporheal zone, the area below the water-substratum interface of streams, is an exciting topic of recent investigation but remains largely undefined. However, it is the site of DOM uptake by microbes and the effects of surface mining coal on this potentially important segment of the stream ecosystem could be significant. Silt might also interfere with the feeding of some macroarthropods by burying food or filling the interstitial spaces of the stream bottom where they forage. Many macroarthropod species have been observed to drift or otherwise try to avoid areas of silt deposition (De March 1976, Ciboroski 1977, Rabeni and Minshall 1977).

It is beyond the scope of this dissertation to advocate either community structure or function as the best approach to the analysis of lotic ecosystems. However, it is intended as a broad based treatment of the topic of surface mining effects on the structure and function of stream biological communities. Those sections dealing with structural properties of stream communities represent the continuation of an established approach to studying this problem. Those areas dealing with functional properties of biological communities represent an attempt to apply new ideas generated by contemporary theory in lotic ecology to the study of this problem.

There are two recent theoretical models in lotic ecology of particular relevance to this dissertation. The river continuum concept of Vannote et al. (1980) provides a model for predicting changes in the functional organization of stream biological communities that result from changes in the physical structure of streams, especially relating to organic matter, that are correlated with increasing stream size. While changes in the physical structure of streams caused by surface mining may be different than the systematic set of changes associated with stream size, the principle that functional organization responds to structural change should still apply. The other model is called the resource spiralling concept and provides a lotic analogue to the nutrient cycling model of terrestrial ecosystems (Webster 1975, Newbold 1981, 1982, Elwood et al. 1983).

The following description of the river continuum model covers those important features which demonstrate its applicability to this study, but is not intended to be a complete treatment of the theory. The river

continuum concept predicts that as stream size increases the relative concentration of FPOM, or at least the ratio of FPOM to CPOM, increases. This occurs for a number of reasons including the fact that smaller streams receive larger amounts of CPOM relative to their surface area (Conners and Naiman 1984) and are more retentive of it (Bilby and Likens 1980, Bilby 1981). The ratio of FPOM increases downstream because much of the POM there is processed CPOM from upstream. The model also predicts that the P/R ratio (photosynthesis/respiration) should increase with stream size. Autochthonous primary productivity is lower in the small shaded upstream reaches of the stream and increases as the canopy opens up downstream. This increased primary productivity also contributes to FPOM of lower stream reaches in the form of unicellular diatoms and algae. Even further downstream primary productivity may again be limited by depth and turbidity resulting in a decrease in the P/R ratio. The central principle of the theory is that stream biological communities should have a functional group composition which reflects their position along this river continuum. Therefore, small forested headwaters should have relatively more shredders than lower reaches, while lower reaches should have relatively more filter-gatherers. Middle reaches should have more grazers. One of the questions posed in this research is whether or not the POM structure of streams is altered and if so, does the functional organization of the macroinvertebrate community reflect it.

The resource spiralling concept provides a potentially useful approach to summarizing the effects of surface mining on streams. The spiralling length of a nutrient or carbon is the average downstream

distance a unit (molecule) would be transported before completing a cycle through the biological and physical compartments of the stream ecosystem. In the case of carbon, spiralling length is inversely proportional to the retentiveness of the stream for POM and DOM, and to the efficiency with which stream biological communities utilize it. Spiralling length could be an useful index for summarizing the overall effect of surface mining coal on the retentiveness and utilization of organic matter in streams. Spiralling length is not directly measured in this study but some of the parameters estimated herein may provide insight into the probable effect of surface mining on this parameter.

While this study does not experimentally evaluate these theories, it shows that they have proven useful in directing the focus of this research because of the ideas they have generated concerning the interaction between the watershed and its stream (and associated biological communities). With the implications of these theories in mind, this dissertation seeks to obtain functional as well as structural explanations of the effects of surface mining on the macroarthropod fauna of streams and, hopefully, to better describe the causal mechanisms involved.

Description of Study Area

The two streams in this study, Crabapple and Bruce Creeks, are located in the Cumberland Mountain section of the Appalachian Plateaus physiographic province (Fenneman 1946) in Campbell County, Tennessee. The U.S. Geological Survey gauging station near the confluence of Crabapple Creek with Louse Creek is located at 36° 27' 25" N, 84° 9' 30" W, 7.4 km NNW of La Follette, Tennessee, on the Ivydell Quadrangle.

Leist et al. (1981) described the terrain of this region as steep with slopes ranging from 20 to 60%. Rocks are of Pennsylvanian age and contain numerous economically extractable coal seams. Soils are mostly acidic, derived from sandstones, siltstones, and shales, and are highly erodable. Average annual precipitation ranges from 46 to 55 inches.

Both streams are first order tributaries of Louse Creek, part of the Clear Fork of the Cumberland River system. The drainage basin of both streams is mostly forested. The forest consists primarily of oak, hickory, and tulip poplar with some hemlock concentrated near the stream. Extensive areas of rhododendron also occur along the stream margins. These two streams were selected for comparative purposes because they are geologically and floristically similar and are located near each other, minimizing climatic differences and facilitating sampling. Crabapple Creek is relatively pristine with no sign of human habitation. Virtually all the streams of this region have experienced some logging and Crabapple is no exception. A new logging operation was being initiated in the summer of 1983, but only road improvements and the removal of an insignificant number of trees had occurred by the end of this study in May 1984.

Bruce Creek is probably similar to Crabapple Creek with respect to logging history. The upper end of Bruce Creek contains two buildings and some small pasture areas. About 3% of the surface area of Bruce Creek watershed has been disturbed by surface mining coal. The last known mining activity, ascertained from aerial photos (Tennessee Valley Authority), occurred in 1967-1968 but mining may have also occurred in earlier years. No reclamation was required or practiced at that time.

The abandoned mine lies on the western boundary of the basin at or slightly above the 1900-ft contour, probably on the Jordan coal seam.

Objectives

The objective of this study is to relate the effects of surface mining coal on the physical structure of streams to the observed differences in the structure and function of stream macroarthropod communities. The macroarthropod community for the purposes of this study includes aquatic insects and the aquatic crustacean Lirceus fontinalis. Other crustacea, decapods, and amphipods (rarely collected), occur in these streams but were not sampled. The structure of the stream macroarthropod communities is described in terms of its taxonomic composition, abundance and diversity. Function of the macroarthropod community is described in terms of its functional feeding group composition and secondary productivity of these groups. Overall function is indirectly assessed by total productivity and leaf processing rates. The distribution and transport of POM will be referred to as simply organic matter dynamics. The organic matter dynamics of the two streams were compared by estimating their input, standing stock, transport, and relative consumption rate of POM. POM distribution in the stream is a function of the physical properties of the stream, but POM is in itself an important physical attribute of the stream environment that provides an energy resource and habitat for many stream organisms. Previous observations of altered taxonomic composition in surface mined streams have suggested that some orders of aquatic insects are more affected than others to post-mining stream conditions. These orders may be physiologically sensitive to stream conditions. However, water of mined streams

usually does not contain toxic levels of any pollutants, except perhaps silt. Therefore an alternate explanation is that some trophic categories (functional groups), which dominate some orders, are more affected than others by the changes in the physical properties of the stream including its POM dynamics. Quantifying the organic matter dynamics of the two streams allows comparison of the overall retentiveness of the two streams and establishes whether the macroarthropods share a similar resource base in both streams.

Overview

Chapter 2 presents data comparing some aspects of the organic matter dynamics in Crabapple and Bruce Creeks. The input, standing stock, some limited aspects of transport, and processing of POM was estimated for both streams. POM particles vary in size from microscopic to the boles of large trees. Only particles ranging from 0.45 mm to small branches were included in this study. POM was separated into CPOM, FPOM, and wood. The input of POM directly from the forest canopy (fall-in) and indirectly from the forest floor (blow-in) was estimated. The standing stock of POM was estimated from Surber samples. The difference between input and standing stock must be due to either consumption or downstream export. Downstream export was not effectively measured but in so far as secondary production estimates consumption, a relative comparison between the two streams is still possible. Consumption was also measured more directly by comparing the rate that leaves were processed. Processing of leaves, the rate at which leaves loose weight in the stream due to physical and biological factors, was compared in the two streams using the leaf bag method of Elwood et al.

(1981). Differences in the POM dynamics between the two streams might help explain differences in taxonomic and functional feeding group composition between the two streams. In so far as POM is an important component of the physical structure of streams the physical similarity of the two streams is being compared. The processing of CPOM is however a functional aspect of the streams in that it depends on the activity of the microbial and macroarthropod communities.

Chapter 3 compares the structure of the macroarthropod communities in terms of the species richness and diversity of the two streams as well as the relative abundances at various levels of taxonomic classification (order, family, genus, species). The functional group composition of the two streams is compared and related to the taxonomic composition. The community structure of the two streams is consistent with expectations based on previous research in this area. The functional group composition of the two streams reveals interesting differences which could go undetected in an analysis limited to community structure.

Secondary productivity of the dominant macroarthropods and functional feeding groups of the two streams are compared in Chapter 4. The relative amount of production in different species and functional feeding groups is another way of assessing functional responses of stream biological communities to the effects of surface mining coal. Better understanding of the functional response of stream communities might lead to reclamation techniques that are more effective in mitigating the adverse environmental effects of surface mining on stream ecosystems.

CHAPTER 2

PARTICULATE ORGANIC MATTER DYNAMICS IN A MINED VERSUS A PRISTINE STREAM IN THE CUMBERLAND MOUNTAINS OF EASTERN TENNESSEE

Introduction

Theory in lotic ecology has been developing along an independent course from the applied disciplines of environmental monitoring and impact assessment. Ever since the determination of organic matter budgets of small streams demonstrated that they are primarily heterotrophic systems (Fisher and Likens 1973, Sedell et al. 1974) lotic ecologists have focused on developing realistic general models to describe them. As much as 99% of the energy input of small streams is detrital carbon from outside the stream (allochthonous) in the form of senescent leaves, wood, and other debris. One conceptual development was the realization that stream biological communities can be subdivided into "functional" groups of species based on similar strategies and adaptations for processing this detrital energy source. Reviews emphasizing the role of aquatic macroinvertebrate functional groups in the processing of stream carbon resources have been written by Cummins (1974), Cummins and Klug (1979), and Anderson and Seddell (1979).

The functional group concept led to the development of a general model of organic matter processing in streams (Petersen and Cummins 1974, Boling et al. 1975a, 1975b). A rapidly expanding empirical base demonstrated that carbon resources and functional group composition of streams vary along a gradient of stream size. These observations have

resulted in continued elaboration of stream carbon models. The functional group organization of streams is now viewed as a dynamic assemblage of trophic strategies that responds to the distribution of carbon resources which is determined by changes in the physical structure of streams. A comprehensive general theory of lotic ecology describing the interactions between the stream and its watershed and the effects on the functional organization of its biological communities has now been proposed. The river continuum concept (Vannote et al. 1980) brings together in one model all aspects of the previously mentioned theories. At present the theory is still being evaluated but results so far have generally upheld its validity (Hawkins and Sedell 1981, Minshall et al. 1983, Conners and Naiman 1984). The river continuum concept holds great promise for directing research and interpreting results over a wide range of topics in lotic ecology including environmental impact assessment. Its potential contribution to studying the impacts of surface mining coal on streams is especially great considering that this impact is mediated through its effect on watersheds which results in physical changes in the structure of the stream.

It was soon appreciated that by a combination of physical and biological processes carbon and nutrients could be retained within streams. This recycling of materials in spite of the obvious flow through nature of the lotic environment has been termed nutrient "spiralling" to distinguish it from its terrestrial analogue, nutrient cycling (Webster 1977, Newbold et al. 1981,1982, Elwood et al. 1983). This theory provides yet another new perspective from which to view the effects of surface mining on stream ecosystems.

The impact of rapid theoretical developments in lotic ecology is yet to be realized in the area of environmental monitoring. This author is aware of only one other study which evaluated the effects of surface mining coal on the physical structure and functional group composition of streams (Matter 1978). Research on some other environmental impacts on streams has begun to focus on functional responses of stream biological communities. Specht et al. (1984) evaluated the community structure and functional group responses of stream invertebrates to fly ash effluents. Larrick et al. (1981) evaluated structural and functional responses of aquatic heterotrophic bacteria to thermal, heavy metal, and fly ash effluents. Webster and Waide (1982) evaluated the effects of clear cutting on leaf breakdown in a southern Appalachian stream. O'Hop et al. (1984) compared the secondary production of a stream shredder, Peltoperla maria, in logged and pristine watersheds.

The objective of the present study is to evaluate the difference, if any, in the organic matter dynamics of a surface mined stream compared to an unmined reference stream. Organic matter, its abundance, distribution, and relative size is an important component of the physical structure of streams. In addition to providing the bulk of the energy base of small forested streams it also serves as habitat for many organisms. The organic matter dynamics of a stream could be affected by changes in other structural properties of the stream resulting from surface mining such as siltation and increased frequency and severity of floods. Differences in the organic matter dynamics of streams should result in differences in the functional organization of their biological

communities and possibly in differences in their biological structure and productivity.

In order to assess potential effects of surface mining coal on the structure and function of lotic ecosystems the organic matter dynamics of two streams were compared. Crabapple Creek, a pristine control stream, and Bruce Creek, a stream with 3% of its watershed surface area disturbed by mining 15 years prior to the study, were compared for inputs, standing stock, transport, and processing of organic matter. Inputs of allochthonous carbon were measured in order to establish the similarity of the two watersheds (floristically) and to compare the energy base of the two streams. A knowledge of relative input of organic matter was essential to comparing the standing stock of POM in the two streams. Standing stock POM was measured and, considered along with estimates of POM inputs, provided a measure of stream POM retentiveness. Transport of particulate organic matter through the streams was compared on the basis of qualitative drift samples and suspended solids analysis of water samples. It would, of course, be highly desirable to measure all the POM transport in the streams, but unfortunately this difficult task was not possible in this study. The processing rate of leaves which are the single most important component of CPOM is an important indicator of the functional organization of the stream. The processing rate for two species of leaves was measured by the leaf bag method of Elwood et al. (1981). Processing rate of leaves reflects the activity of microbial conditioners and macroarthropod shredders, two important functional groups in lotic ecosystems. The above measurements

yielded a partial organic matter budget for the two streams and provided insights into their structural and functional differences.

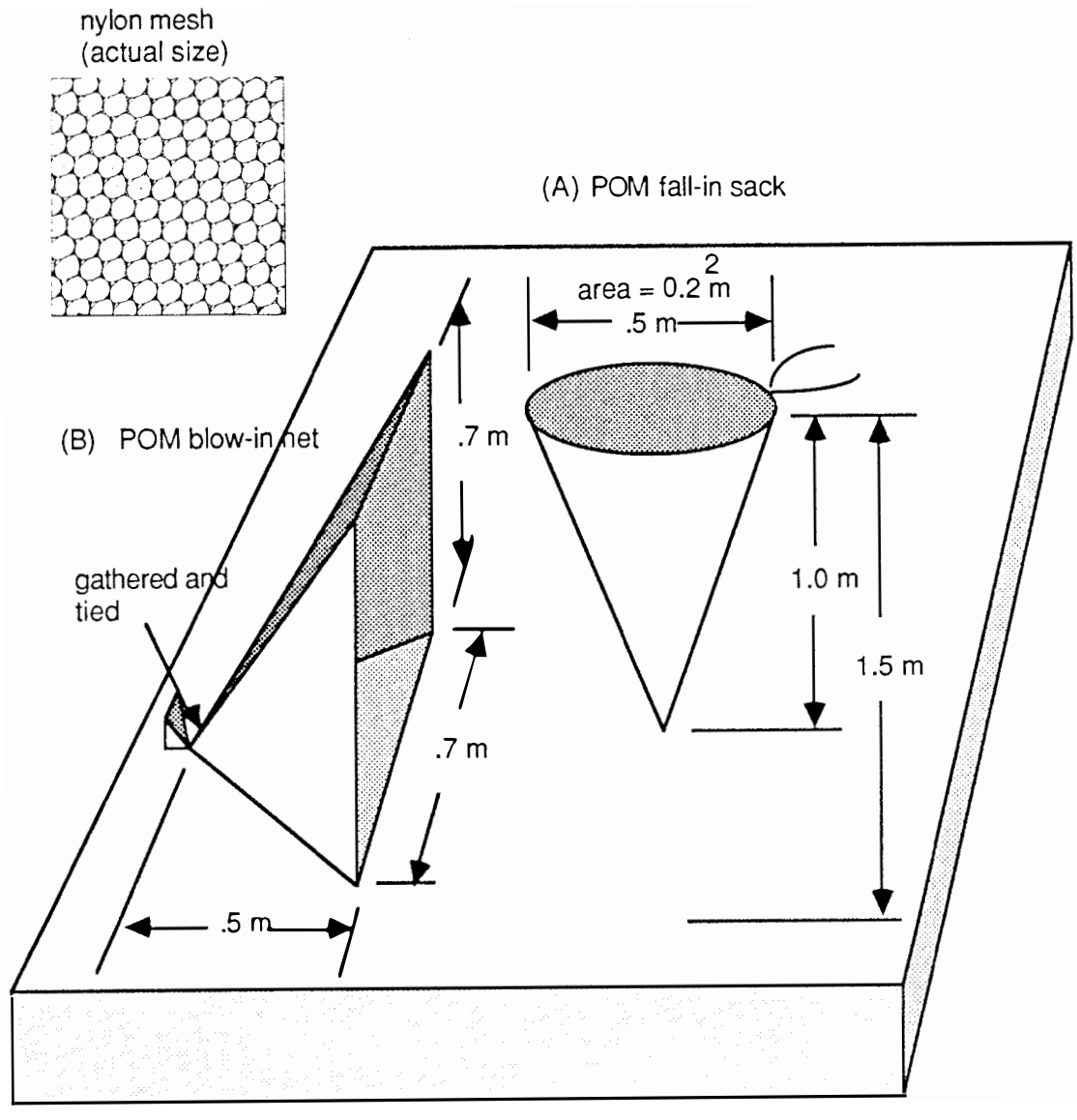
Materials and Methods

Physical and chemical water properties were measured in each stream on 25 dates over a 30 month period. Water temperature, dissolved oxygen, pH, alkalinity, hardness, SO_4 concentration, Fe concentration, and turbidity were measured in the field whenever possible. Water parameters were measured in the lab on water samples collected for suspended solids analysis when field readings could not be obtained. Water temperature and dissolved oxygen were initially measured with a Yellow Springs Instruments dissolved oxygen meter. Water temperature was measured with a thermometer after dissolved oxygen measurements were discontinued because the streams were always very similar and always near saturation. The pH was measured with an Orion pH meter. Field pH measurements can be difficult despite the claims of the many manufacturers of this technology. Instrument readings often varied inexplicably between measurements or drifted to a disturbing degree during measurements. Dyer (1983) discontinued field pH measurements with a Markson digi-sense pH meter, model 5985-40, because of inconsistent readings. Field pH readings were also difficult in this study but were nevertheless measured for the duration. U.S. Geological Survey research conducted on Crabapple Creek provide some independent comparative values. No extreme pH measurements were recorded despite the inconsistent readings. Alkalinity, hardness, and dissolved iron and SO_4 concentrations were measured according to Standard Methods (American Public

Health Association [APHA] 1975) with a Bausch and Lomb Mini Spectronic 20 and Bausch and Lomb Spectrokits.

Water samples for suspended solids analysis were collected in 250 ml Nalgene bottles and analyzed in the lab for suspended and volatile fractions by Standard Methods (APHA 1975). The volatile fraction was assumed to be very fine suspended POM. The minimum particle size retained in this analysis was 0.45 µm. Very little of the material collected on the filters was visible and probably did not significantly overlap the larger size range of fine particles collected in Surber samples for standing stock estimates or drift nets for POM transport estimates.

Input of allochthonous carbon was estimated with two types of sampling devices (Figure 2.1). Litter falling directly into the stream, fall-in, was captured in conical fabric sacks. The mouth of the sack was .5 m in diameter, with an area of approximately 0.2 m^2 The sacks were of muslin or 100% rayon suspended from a loop of 10 gauge wire. The ends of the wire were twisted around narrow saplings in order to suspend the sacks approximately 1.5 m above the ground or stream. Sixteen sacks were located randomly along each stream channel. Litter blowing or rolling downhill into the stream, blow-in, was intercepted in nets of coarse nylon fabric with octagonal 3 mm mesh openings. The net was supported by a frame of 10 gauge wire and the front and back were secured with 10 gauge wire stakes. The net mouth was 0.7 m by 0.7 m (approximately 0.5 m^2) and opened away from the stream channel on bench areas or upslope at the base of slopes. The nets were deployed to provide a stratified random sample of each stream, with 12 nets placed



(A) POM sack for sampling fall-in of organic matter.
(B) POM net for measuring blow-in of organic matter.

Figure 2.1. Sampling devices used to measure POM inputs to Crabapple and Bruce Creeks.

at the bases of slopes (30-60%) and 12 at the edge of the stream in areas with flood-plain benches. The nets were allocated equally to both sides of the stream. Sacks and nets were emptied approximately monthly for 13 consecutive months.

Sacks were assumed to sample all POM fractions: coarse particulate organic matter [CPOM], fine particulate organic matter [FPOM], and wood. Nets were assumed to sample only CPOM and wood, reflecting the implicit assumption that FPOM moves negligible distances after falling to the ground. The POM collected in fall-in sacks and blow-in nets was returned to the lab and frozen for later processing. In this study CPOM was any leafy organic particle retained in a 1 mm US Standard sieve. FPOM is the fraction passing through this sieve, except in the analysis of standing stock of POM in the stream, where FPOM collected in the Surber samples was assumed to be smaller than 1 mm but larger than .56 mm, the mesh size of the Surber net. Wood was mostly small twigs, although an occasional large stem landed in or on the sack, in a net, or under a Surber sampler (only that part transecting the sampling device was kept). Soft winged seeds of maple and tulip poplar were considered CPOM, while mast crops such as beech and hickory nuts and acorns were counted as wood.

Standing stock of POM in the streams was estimated from standard Surber samples (0.1 m^2). POM and macroinvertebrates were separated from sand and gravel by sucrose flotation (specific gravity = 1.12) (Pask and Costa 1971). The macroinvertebrates were then separated from POM by hand picking. (Note: the macroinvertebrate composition of the two streams is discussed in Chapter 3). The organic matter fraction (detri-

tus) was then separated into CPOM and FPOM by wet sieving. The organic matter was sifted three times through a 1-mm US Standard sieve by gently agitating the contents of the sieve in water. The retained CPOM was removed from the sieve and the FPOM passing through was collected by filtration (qualitative 0.45-µm filter paper).

Relative transport of POM in the two streams was estimated from drift net samples. The net mouth was 30.5 cm high by 45.75 cm wide and the mesh size was .36 mm. The nets were placed at narrow points in the stream channel in order to sample as large a fraction of the total stream flow as possible. The total volume of water passing through the net was not measured so the estimates are only useful for comparing the relative amounts between streams. The nets were left in place for approximately one hour at each sample location. Two drift samples were obtained from each stream on most sampling dates. POM was processed in the same manner as described for Surber samples. The amounts of FPOM reported should be considered minimal estimates because it was difficult to remove all of it from the drift net. This FPOM was different from the FPOM collected on filters in the suspended solids analysis. While particles down to 0.45 µm. were retained by glass filters the particles retained by drift nets were no smaller than 0.36 mm. All organic matter samples were dried for at least 24 hours in a drying oven at 100^{o} C and then weighed immediately to the nearest 0.01 g on a Mettler electric balance.

Leaf processing rates were estimated for leaves of two species, beech and flowering dogwood, by placing preweighed amounts of leaves in coarse nylon mesh bags in the two streams on the same date and then

removing several bags from each stream at intervals (Elwood 1981). The nylon mesh is the same as that used to construct the POM nets. The fabric was sewn into bags using heavy duty nylon thread. Leaves were placed inside the bags and the ends were sewn closed so that a loop of fabric at each end could be used to secure the bag to the stream bed. The bags were tied to rocks with nylon cord and placed in riffle sections of the stream. A total of 41 bags of beech leaves and 48 bags of dogwood leaves were recovered from the streams. The bags were removed from the stream, placed in plastic bags, and stored frozen until processed. Leaves were rinsed of silt and dried at 100 ^{o} C for 24 hr and then weighed to the nearest .1 g.

The data for POM inputs were analyzed as a paired sample. This approach was taken because the sampling devices were placed in permanent locations for the duration of the experiment. Permanent placement of sampling devices leads to correlation of the amounts of POM collected from date to date, violating assumptions about homogeneous and independent variances to which analysis of variance is sensitive. All paired T-tests were performed on the means with the null hypothesis being that the differences equal zero, using SAS (Statistical Analysis System) procedure, MEANS (SAS Institute Inc. 1982). POM in drift nets was also analyzed by a paired T-test since there were two replicates per stream per date. Standing stock and total and volatile suspended solids were compared by analysis of variance using SAS, procedure GLM (General Linear Models). Because only two means were being compared the analysis was essentially a T-test for between stream differences; however, the use of a two way factorial ANOVA model (stream by sample date) allowed

the effects of differences between dates to be assessed. It also included a much larger number of error degrees of freedom. Leaf bags were compared in an analysis of covariance also using SAS, procedure GLM (SAS Institute Inc. 1982, Freund and Littell 1981).

Results and Discussion

Water quality data for Crabapple and Bruce Creeks are presented in Appendix A, Table A1. The water quality of the two streams was compared in a paired T-test of six key water quality parameters. Most samples are the average of two measurements per stream on the same day. Table 2.1 presents the number of samples, mean difference, T value, and significance level.

Table 2.1. Paired T-test of water quality parameters between Crabapple and Bruce Creeks.

Variable	N	Mean difference	T	Pr>|T|
pH	21	-0.02	-0.16	0.8734
alkalinity	24	-0.02	-0.07	0.9426
sulfate	23	56.13	18.46	0.0001
iron	24	0.15	1.22	0.2353
hardness	24	59.56	12.20	0.0001
turbidity	24	15.90	4.20	0.0003

Bruce Creek displays the telltale characteristics of a surface mined stream. The elevated sulfate concentration, hardness, and turbidity are all characteristics strongly correlated with mining (Minear et al. 1980). Sulfate concentrations in unmined watersheds are generally low (1 to 20 mg/l), while concentrations in mined watersheds are vari-

able but may be very high (20-200 mg/l)(Dyer and Curtis 1977, Dyer 1982). The insignificant difference in iron concentration is a little surprising but may reflect the relatively small area, approximately 3%, of Bruce Creek disturbed by mining and/or the generally low flow levels under which it was sampled. Most of the iron in stream water is suspended, not dissolved (Leist et al. 1982); therefore, the concentration would be increased by higher flows. The pH and alkalinity levels were about the same although pH measurements were often inconsistent in the field. However, when pH was measured in the lab, the streams were still similar. It is not uncommon for pH and alkalinity to increase after surface mining in this region. Although Bruce Creek clearly has reduced water quality compared to Crabapple Creek, the values are moderate compared to many other surface mined streams (Vaughan et al. 1982). The turbidity values would probably have been much greater in Bruce Creek had it been sampled on random days which would have included higher discharges, but because these streams were difficult to reach they were sampled mostly at lower flows. The average of values over all sample dates are presented with data from Minear et al. (1980) and Williams (1981) in Table 2.2 to illustrate the range of water quality values typical of similar small streams in east Tennessee.

In comparing these data to Crabapple and Bruce Creeks it should be kept in mind that the other streams were accessible on most days from nearby roads so they were sampled over a wider range of weather and flow conditions. Sulfate concentration is often inversely related to stream flow while iron concentration and turbidity increase with stream flow (Leist et al. 1982). Despite these sources of variation a clear differ-

Table 2.2. Averaged water quality data from the present study, and from Minear et al. (1980) and Williams (1981) for several small streams in eastern Tennessee.

Stream	History [1]	pH	Alk	Hrd	SO_4	Fe	Trb
			mg/l				ftu [2]
		Present study					
Crabapple	unmined	7.2	19.6	20.8	0.7	0.1	5.3
Bruce	15 yrs, 3%	7.1	19.4	80.4	57.4	0.3	19.1
		Minear et al. 1980					
Lowe	unmined	6.6	9.6	13.1	8.8	0.06	nr
Anderson	0 yrs, 9%	7.1	41.0	64.0	18.4	0.5	nr
Bill	5 yrs, 11%	6.8	13.0	65.0	51.8	0.1	nr
Green	5 yrs, 24%	6.7	32.8	155.6	129.0	0.07	nr
		Williams 1981					
Lowe	unmined	6.5	8.0	11.4	7.0	0.04	nr
Indian	unmined	6.8	7.0	11.4	7.0	0.06	nr
Dry	25 yrs, nr	6.8	10.0	14.0	6.4	0.06	nr
Duncan	25 yrs, nr	6.6	19.0	62.0	61.0	0.04	nr

[1] years since last mined, per cent of watershed area disturbed

[2] ftu = formazin turbidity units (assumed equivalent to both nephelometric and Jackson turbidity units)

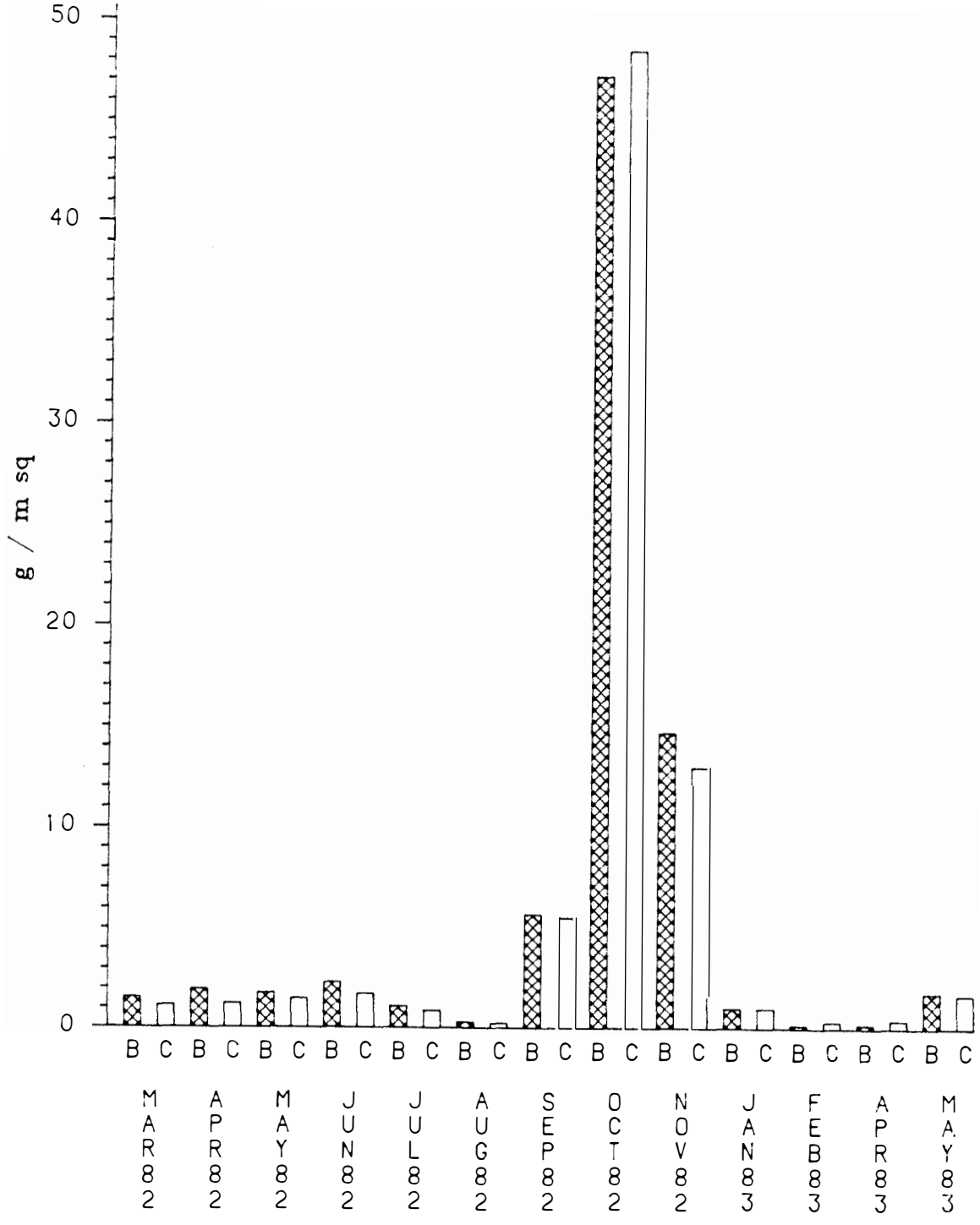
nr = not reported

ence between the mined and unmined streams is apparent, except for the moderate water quality parameters in Dry Creek which indicate substantial recovery of water quality 25 years after mining.

Fall-in data from POM sacks is presented in Appendix A, Table A2. Figure 2.2 charts the monthly amount of CPOM, FPOM, and wood collected in the POM sacks for each stream. Each parameter had a characteristic input distribution throughout the year. Peak fall-in of leaves occurred in October and almost all fall-in occurred between August and December. FPOM input was more evenly distributed but peaked in August with a second lesser peak in October. The August peak may include a large component of insect frass while the October peak may be smaller particles associated with the falling leaves. The fall-in of wood was more erratic, reflecting the more stochastic nature of branch and twig loss from trees. The largest fall-in of wood occurred in April perhaps associated with windy weather and the increased weight of new leaves. A paired T-test of the monthly average values for CPOM, FPOM, and wood was performed. The average difference between means, T-value, and significance level (Pr>|T|) is presented below.

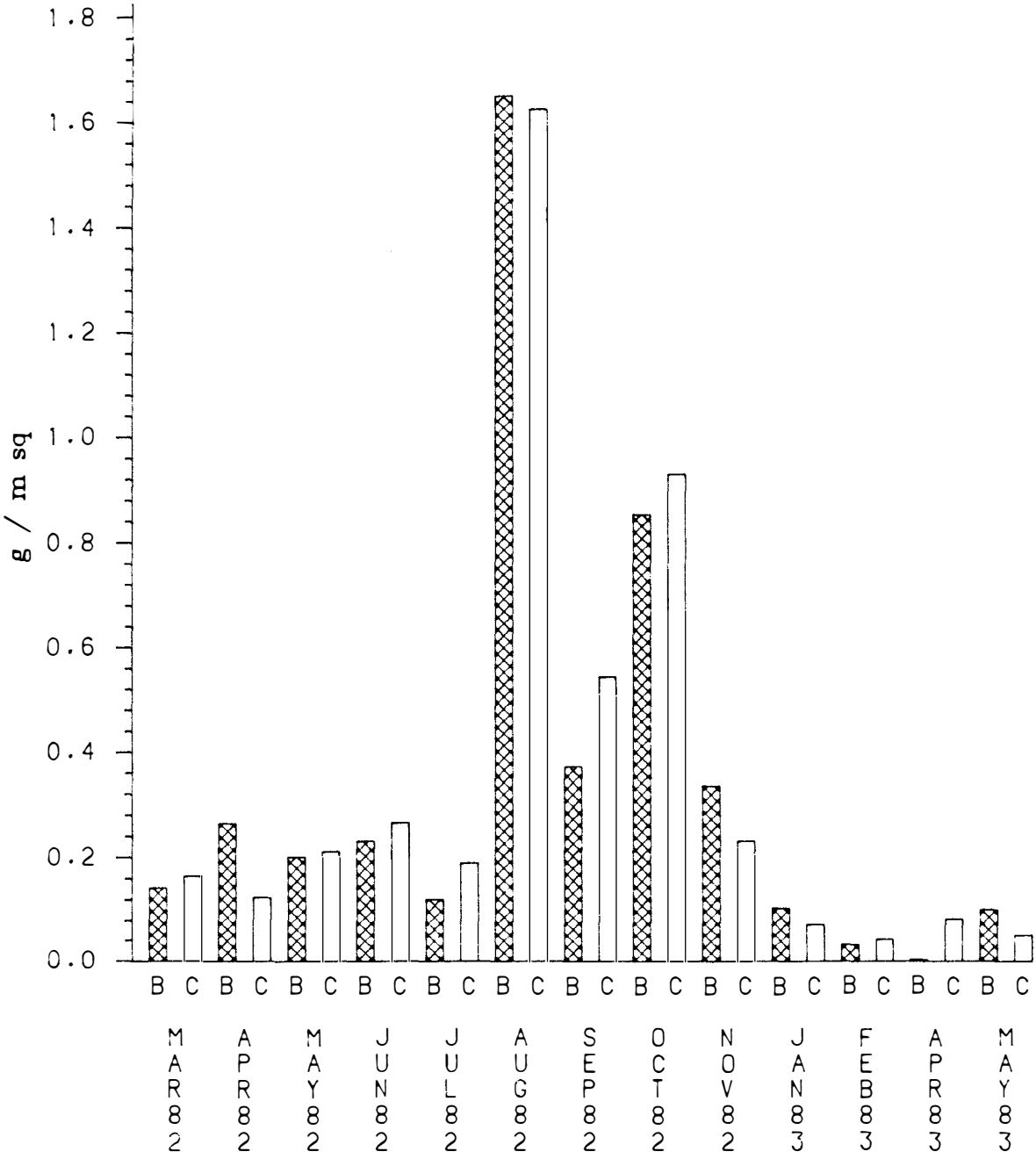
Variable	Mean difference	N	T	Pr>|T|
CPOM	-0.20	13	-1.06	0.3087
FPOM	0.01	13	0.41	0.6920
wood	-0.27	13	-0.70	0.4969

These samples do not reveal statistically significant differences between Crabapple and Bruce Creeks in inputs of any component of POM. These data do not prove that the forest composition or canopy development of the two streams is the same but if any differences exist they



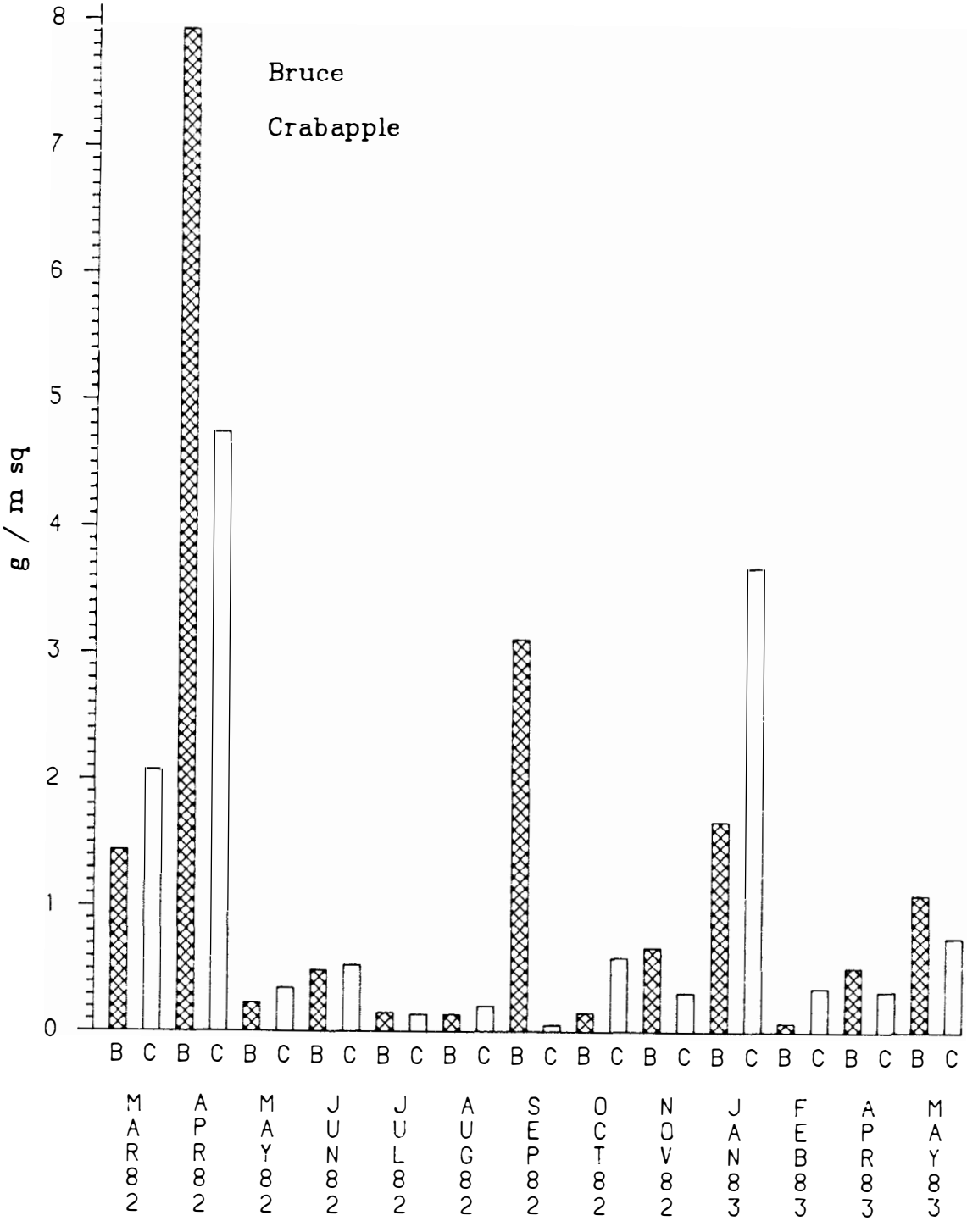
(A) CPOM: coarse particulate organic matter (> 1 mm).

Figure 2.2. Average fall-in of POM collected in Bruce (B) and Crabapple (C) Creeks on 13 dates from March 1982 to May 1983.



(B) FPOM: fine particulate organic matter (< 1 mm).

Figure 2.2. (continued).



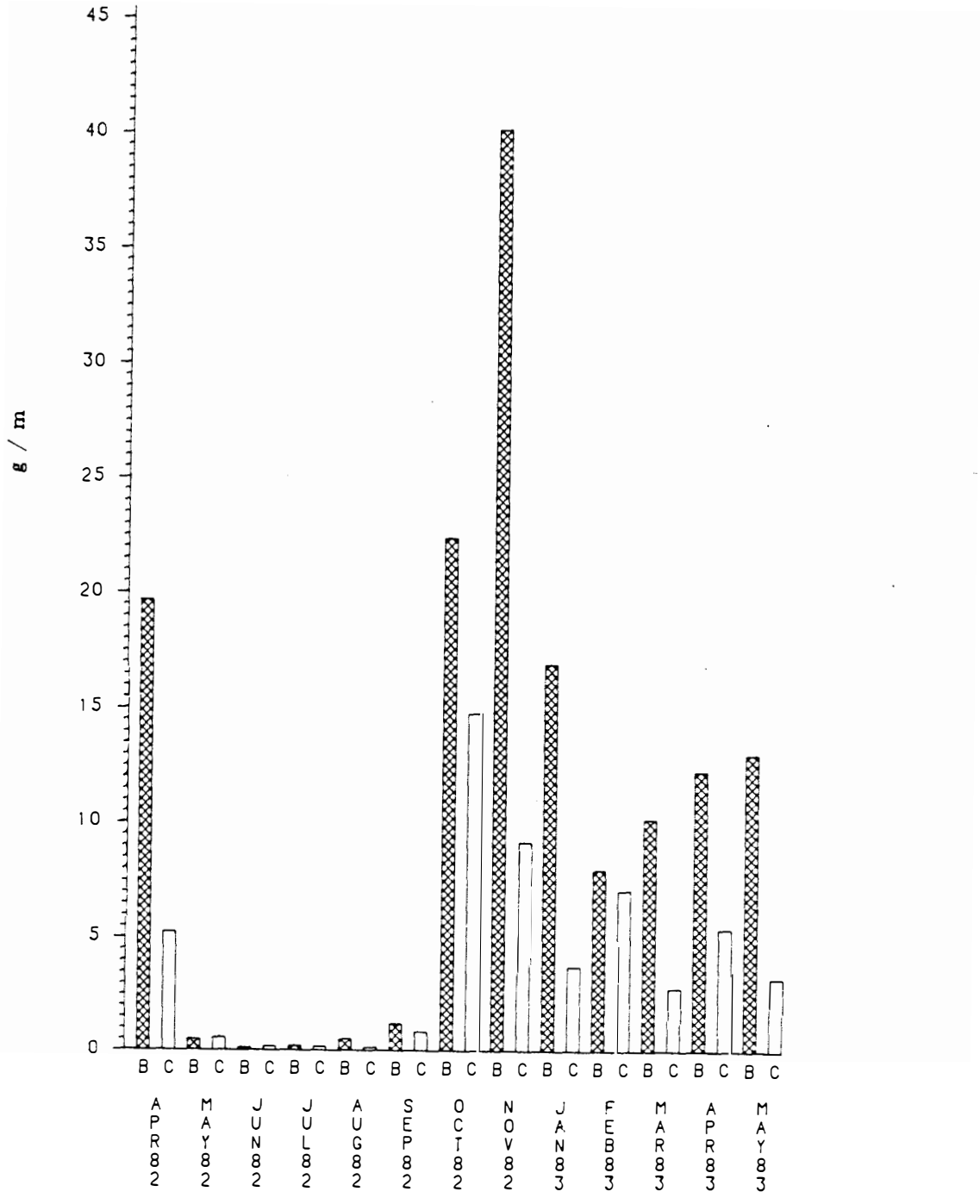
(C) Wood: twigs (< 5 cm), woody mast, and cones.

Figure 2.2. (continued).

are not reflected in the POM estimates. The two watersheds seem to contribute similar amounts of litter directly to the forest floor and stream channel.

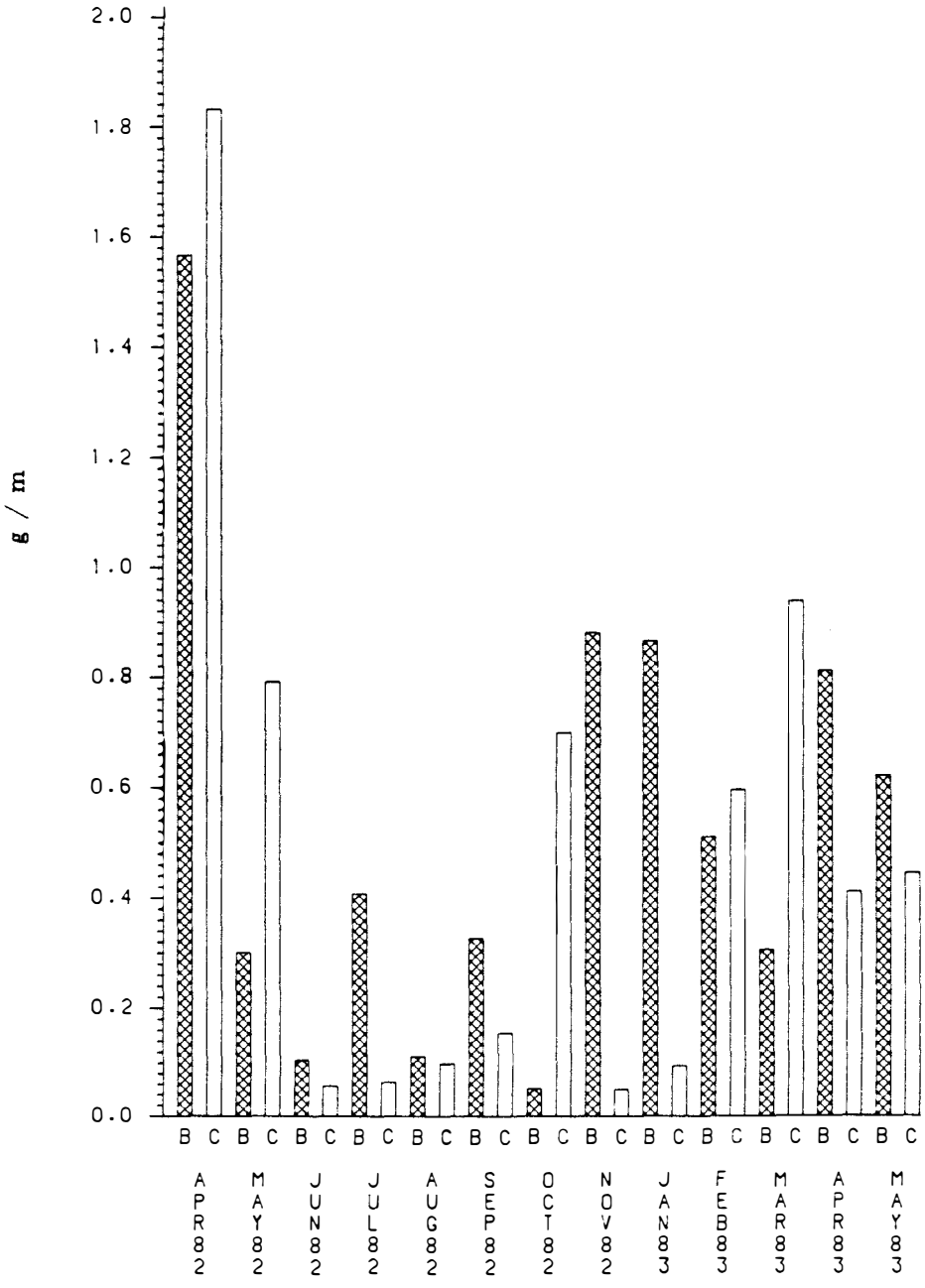
Values for blow-in of CPOM and wood in each stream are presented in Appendix A, Table A3. Figure 2.3 charts the monthly values of CPOM and wood collected along the two streams by the POM nets on benches. Blow-in of FPOM was not measured because it was assumed that most FPOM would quickly settle deep into the litter of the forest floor where it would be protected from further windblown transport. Figure 2.4 charts the monthly values of CPOM and wood collected along the two streams by the POM nets at the bases of slopes. Input of CPOM by blow-in was distributed more evenly during the year than fall-in. CPOM blow-in peaked during or shortly after the peak due to fall-in. Blow-in remained fairly high until April when canopy closing reduced wind velocities reaching the forest floor and a lush, new herbaceous level developed to further limit the movement of leaves. Wood did not move much laterally but input by blow-in was distributed similarly to CPOM. Blow-in data, CPOM and wood, were also compared by a paired T-test. The mean values for 7 to 12 samples per sample date were compared for 13 dates. The results are presented below.

Slope	Variable	Mean difference	N	T	Pr>|T|
bench	CPOM	-10.08	13	-2.85	0.0145
	wood	-0.07	13	-0.37	0.7204
slope	CPOM	-6.89	13	-1.16	0.2679
	wood	-2.35	13	-1.24	0.2386



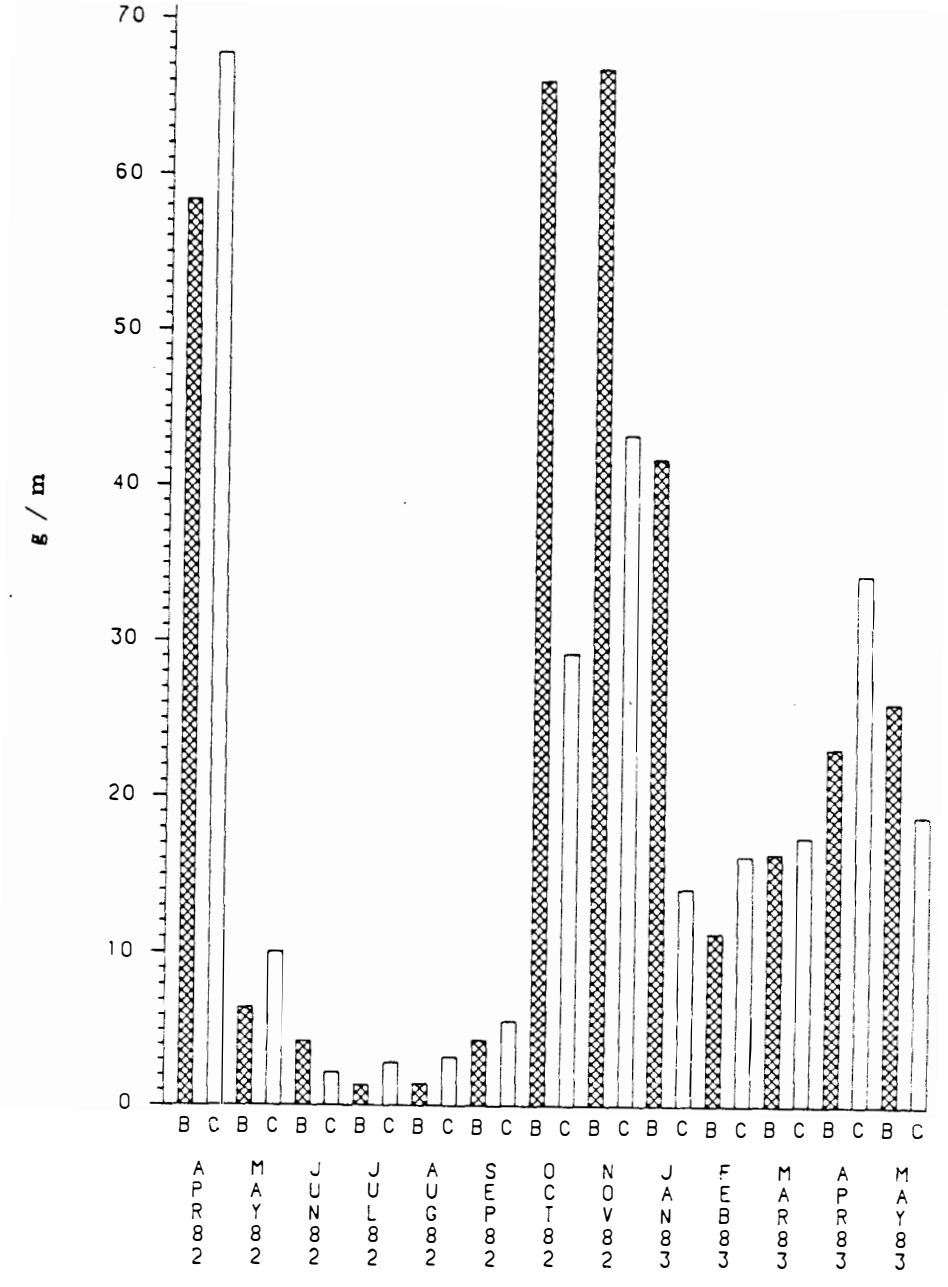
(A) CPOM: coarse particulate organic matter (> 1 mm).

Figure 2.3. Average blow-in of POM along benches of Crabapple (C) and Bruce (B) Creeks on 12 dates from April 1982 to May 1983.



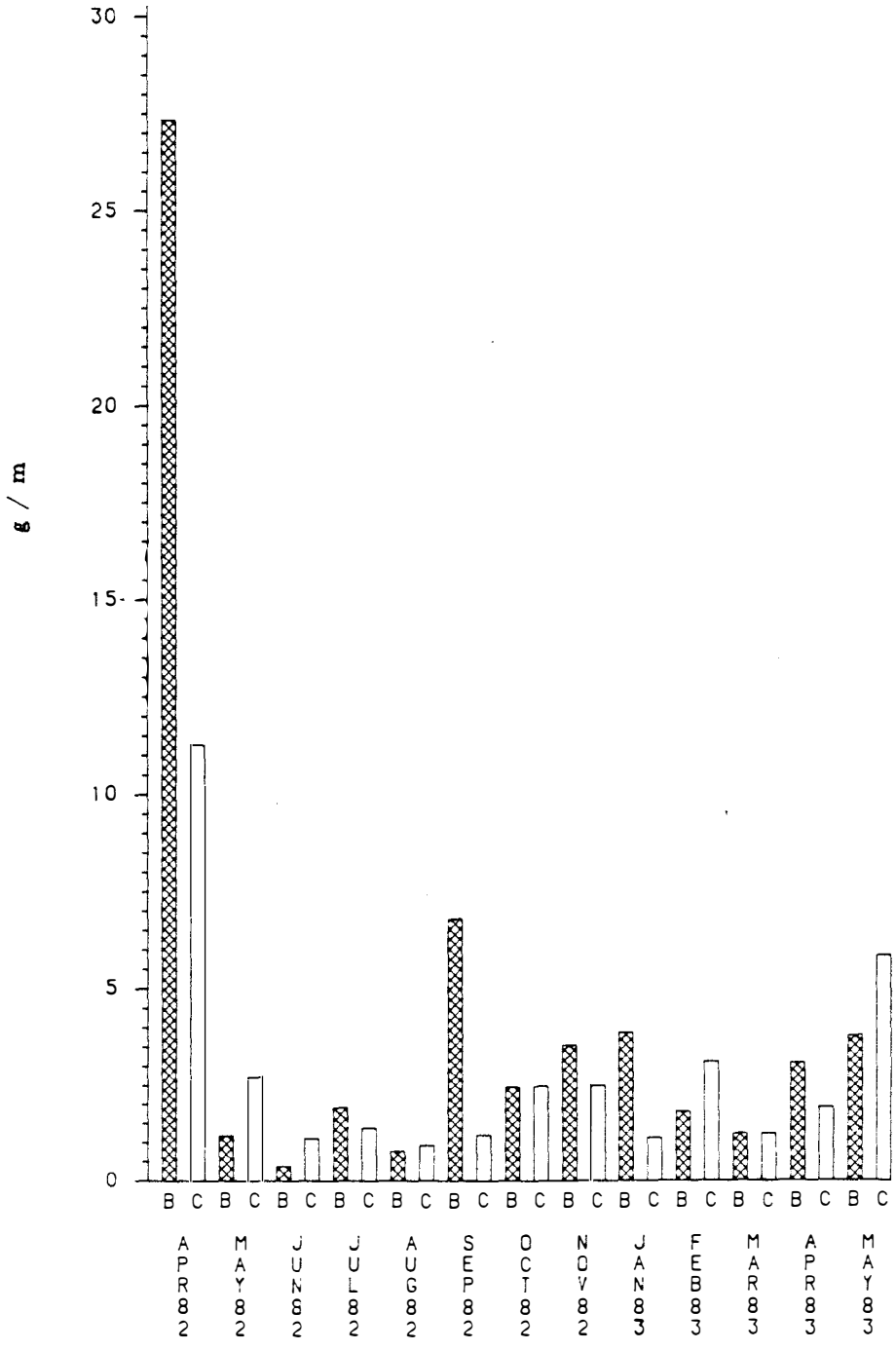
(B) Wood: twigs (< 5 cm), woody mast, and cones.

Figure 2.3. (continued).



(A) CPOM: Coarse particulate organic matter (> 1 mm).

Figure 2.4. Average blow-in of POM along slopes of Crabapple (C) and Bruce (B) Creeks on 12 dates from April 1982 to May 1983.



(B) Wood: twigs (< 5 cm), woody mast, and cones.

Figure 2.4. (continued).

The blow-in of POM was consistently higher in Bruce Creek. However only the blow-in of CPOM from flat benches is significantly different between the two streams. Perhaps the topography of Bruce Creek is more conducive to transport of POM across benches. The benches of Bruce Creek were narrower and three traps had to be placed on bench areas approximately 10 to 20 m above the stream channel where, perhaps, winds were stronger. Table 2.3 compares litterfall and blow-in values for this study to other values reported for southern Appalachian forest streams. Fall-in data are reported in area units (m^2) but blow-in values are reported in length units (m), for meters of stream bank. A novel design was used for the sampling device (Figure 2.1, p.32) used to sample blow-in and its efficiency was unknown. Values for blow-in of litter from this study compare favorably with those for other southern deciduous forest streams verifying the sampling efficiency of this device. The values for all POM inputs to these two streams compare favorably to the results obtained from other streams in southern deciduous forests.

All of the studies presented in Table 2.3 differ in details of sampling technique and processing of samples. Webster and Waide (1982) sampled leaves only, while Webster and Patten (1979) sampled leaves and wood < 5mm, and Comiskey (1978) sampled leaves, wood, fruits, and insect frass as well. In this study fruits were separated into wood and CPOM because this corresponds to functional differences in the utilization of POM by stream organisms. Also insect frass was separated into CPOM and FPOM. The discrepancy between the sum of CPOM and wood and the Total columns for Comiskey's data in Table 2.3 reflects the value of fruits

Table 2.3. Fall-in and blow-in values for the present study and for some other southern Appalachian deciduous forest streams.

Study site stream	CPOM	FPOM	Wood	Total	Reference
		Fall-in (g/m^2)			
Campbell Co, TN					Present study
Bruce Creek	388.1	21.6	80.3	490.0	
Crabapple Creek	374.9	22.1	65.2	462.2	
Coweeta, NC					Webster & Waide 1982
Big Hurricane Branch	259.2	nr	nr	259.2[1]	
Watershed 18					Webster & Patten 1979
year 1	nr	nr	nr	353.2	
year 2	nr	nr	nr	362.0[2]	
Anderson Co., TN					Comiskey 1978
Walker Branch	372.0	nr	26.3	459.1[3]	
		Blow-in (g/m)			
Campbell Co, TN					Present study
Bruce Creek					
bench	119.8	nr	5.4	125.2	
slope	423.7	nr	58.2	481.9	
Crabapple Creek					
bench	44.7	nr	5.4	49.8	
slope	265.4	nr	35.4	300.7	
Coweeta, NC					Webster & Waide 1982
Big Hurricane Branch	174.8	nr	nr	174.8[1]	
Anderson Co., TN					Comiskey 1978
Walker Branch	240.1	nr	11.8	316.7[3]	

[1] leaves only.
[2] includes wood < 5 mm in diameter.
[3] fruits and frass divided between CPOM, FPOM and wood in this study
nr = not reported

and frass which he measured. In addition, the values of blow-in reported by Comiskey (1978) and Webster and Waide (1982) are weighted averages to reflect the relative amount of stream bank in each slope category. Slope categories have been reported separately. Taking into account these differences increases the similarity of the POM estimates for these streams.

The standing stock of POM in Crabapple and Bruce Creeks was estimated by separating and weighing the organic matter collected along with macroarthropods in standard Surber samples. The results of the two way factorial ANOVA comparing the streams are presented in Table 2.4. The type I sums of squares, F values, and probability levels for these F values are shown. Average values of CPOM, FPOM, and wood collected in Surber samples are presented in Appendix A, Table A4.

Crabapple Creek was significantly different from Bruce Creek in standing stock of FPOM only. Bruce Creek samples contained considerable amounts of coal and coal is similar in density to the FPOM in the samples making it difficult to separate them by floatation. After picking out the macroinvertebrates and separating the POM into CPOM and FPOM, an effort was made to exclude any remaining mineral fraction, including coal, from the FPOM. This was a small amount, mostly of sand in Crabapple Creek samples, but often a considerable amount of fine coal particles in Bruce Creek samples. The greater amount of FPOM in Crabapple Creek may have resulted from a systematic loss of FPOM from Bruce Creek samples when trying to separate fine coal particles from organic particles in Bruce Creek samples. The significant interaction term for stream by date indicates FPOM differences were related to a factor which

Table 2.4. Results of the two way (stream by date) factorial ANOVA for average standing stock of POM in Crabapple and Bruce Creeks.

Source	df	Type I ss	F value	Pr > F
		CPOM		
stream	1	65.147	2.05	0.1544
date	8	116.464	0.46	0.8862
stream by date	8	238.321	0.93	0.4898
		FPOM		
stream	1	2.903	35.49	0.0001
date	8	2.758	4.22	0.0001
stream by date	8	2.577	3.94	0.0002
		wood		
stream	1	2.828	0.39	0.6027
date	8	45.923	0.79	0.5595
stream by date	8	80.521	1.39	0.1999

varied with the sampling dates. This may reflect differing amounts of FPOM collected in one of the streams at different flow volumes over time, or, it may reflect increased amounts of coal particles collected in Bruce Creek at higher flow volumes and a corresponding loss of more FPOM in the separation process. The average standing stocks (g/m^2) of each component of POM are presented below.

	CPOM	FPOM	wood
Crabapple Creek:	27.8	4.4	13.4
Bruce Creek:	18.1	2.3	15.0

The structural changes known to occur in surface mined streams, siltation and increased peak flows, could lead to changes in the standing stock of POM in the stream. POM is also part of the physical structure of the stream from the perspective of the macroinvertebrates which utilize it for food and habitat. Siltation could entrap a large amount of POM increasing its standing stock. At the same time, however, much of this entrapped POM might be unavailable for processing by aquatic macroinvertebrates, further increasing its standing stock. Matter (1978) found greater standing stock of POM in surface mined compared to unmined streams in southwestern Virginia. On the other hand the more frequent and severe flood events of surface mined streams might remove debris dams from the stream resulting in lower standing stock of POM. Bilby and Likens (1980) found that the standing stock of POM contained in debris dams was greatest in first order streams and decreased with stream order (size). Over 75 and 58% of the POM of first and second order streams, respectively, was contained in debri dams. Further, the removal of debris dams resulted in dramatic increases in the transport of POM and DOM through the stream (1981). Since Bruce Creek is a slightly smaller stream than Crabapple Creek it might therefore be expected to have a larger standing stock of POM. However, Bruce Creek could have a higher density of macroarthropod shredders which convert the CPOM to a more transportable FPOM faster than occurs in Crabapple Creek. If this were the case, Bruce Creek might actually have a shorter spiralling length of carbon than Crabapple Creek because much of the carbon would be going into the secondary production (biomass) of macroinvertebrates. Other more subtle factors might also be operating. For

example, surface mining might result in higher stream concentrations of nitrogen and phosphorous creating an enrichment effect that accelerates the processing rates of some leaf species. However, a definitive resolution of this question should be based on accurate measurement of POM transport through the stream.

The amount of POM in transport was estimated for both streams with drift nets. It is important to note that these POM transport values represent a very limited range of transport conditions which occur in the streams. This is because the streams were usually sampled at moderate to low discharges and probably during declining phases of the stream hydrographs. While these are the conditions which predominate in the streams, and may therefore be biologically relevant, most of the total POM transport occurs during brief intervals of high flow. Comiskey (1978) observed that peaks in POM transport generally precede peaks of discharge in Walker Branch, Tennessee, although exact dynamics of each storm event vary depending on numerous factors. Comiskey noted that 52% of the annual FPOM transport occurred during a single storm which accounted for 9% of the annual discharge. This was an extreme storm, but the pattern is probably similar in most years. Such dramatic transport events can have profound effects on the standing stock of POM in the stream. The standing stock of CPOM (leaves only) in Walker Branch was reduced 95% by this one storm.

A paired T-test compared the measured values between streams on each date. The average difference for each parameter, number of

samples, T value, and probability level are presented below.

POM fraction	Mean difference	N	T	Pr>|T|
CPOM	-0.185	24	-0.69	0.4961
FPOM	-0.037	24	-0.79	0.4368
wood	-0.045	23	-1.16	0.2565

The average amounts of CPOM, FPOM, and wood collected in the two drift samples for each stream are presented in Appendix A, Table A5. On some dates only one sample was obtained from one or both streams. The POM transport estimates from drift nets presented here are not reliable indicators of relative differences between the streams because the samples can only be considered qualitative since it was not possible to capture the entire stream flow in the nets. There were no significant differences in the transport of POM as measured under these conditions.

The average total and volatile suspended solids for the two streams are presented in Appendix A, Table A6. Total and volatile suspended solids were compared by two way (streams by dates) factorial ANOVA (Table 2.5).

Estimates of total and volatile suspended solids were significantly greater for Bruce Creek than Crabapple Creek. Increased suspended solids are characteristic of streams draining strip mined watersheds. If one assumes that the volatile fraction of the suspended solids is very fine particulate organic matter then Bruce Creek has greater transport of this very fine suspended POM through the stream. However, the volatile fraction of suspended solids in Bruce Creek may contain a large fraction of coal. Therefore these data should be interpreted cautiously. Coal particles are probably slightly denser than most POM,

Table 2.5. Two way (stream by date) factorial ANOVA for total and volatile suspended solids in Crabapple and Bruce Creeks.

Source	df	Type I ss	F value	Pr > F
		total		
stream	1	25173.965	202.40	.0001
date	17	27528.766	13.02	.0001
stream*date	17	21374.197	10.11	.0001
		volatile		
stream	1	317.557	58.67	.0001
date	17	1333.912	14.5	.0001
stream*date	17	570.270	6.2	.0001

but the difference may not be enough to prevent its contamination of the samples. Interestingly, the volatile fraction of suspended solids in Crabapple Creek was approximately half the total amount while in Bruce Creek it was less than one fifth. This indicates that while a larger fraction of the total suspended solids in Bruce Creek was not volatile, it still had a significantly larger amount of volatile suspended solids. The ratio of approximately 50:50 total to organic POM should be typical of small undisturbed streams. This ratio could have differed if the stream were sampled on random dates and the samples had included storm events.

Processing of dogwood and beech leaves was compared in the two streams by an analysis of covariance. The slope of the covariate, i.e. weight loss, corresponds to the processing rate of leaves in the stream.

Previous leaf processing studies have established that an exponential decay model of the form:

$$Y_t = Y_o \, e^{-kt}$$

where: Y_t = the weight of leaves remaining at time t

Y_o = the initial weight of leaves

k = the processing rate of the leaves

t = the number of days in the stream

best describes the weight of leaves remaining as a function of exposure time in the stream (Elwood et al. 1981). This function can be linearized by a log transformation to:

$$\ln(Y_t) = \ln(Y_o) - kt$$

and analyzed by least squares linear regression (Snedecor and Cochran 1967, Elwood et al. 1981). Therefore the analysis was performed on the natural log of the percent of weight remaining on the sampling date in order to linearize the data. A summary of the regression results is presented below:

Leaf	Stream	Processing rate	Std error	R-square
dogwood	Crabapple	-0.0458	0.0093	0.55
	Bruce	-0.0904	0.0118	0.70
beech	Crabapple	-0.0022	0.0006	0.42
	Bruce	-0.0021	0.0003	0.73

Processing of beech and dogwood leaves occurred at very different rates. Beech is known to be one of the slowest species to be processed in streams. Dogwood in contrast is processed very fast. An analysis of covariance was performed on each leaf species to test for homogeneity of the regression (i.e., processing) coefficients in the two streams using

SAS procedure GLM. The Type I sums of squares, degrees of freedom, F values, and associated probability levels are presented in Table 2.6 below.

Table 2.6. Analysis of covariance results for leaf processing in Crabapple and Bruce Creeks.

Source	df	Type I ss	F value	PR > F
		dogwood		
stream	1	3.787	7.46	0.0090
date	1	36.967	72.81	0.0001
stream*date	1	4.442	8.75	0.0049
		beech		
stream	1	0.011	7.96	0.0077
date	1	0.056	39.29	0.0001
stream*date	1	0.000	0.0	0.9661

The significant interaction term (stream*date) for dogwood leaves indicates the regression lines, i.e. processing rates, are different between the two streams. The rate of processing in Bruce Creek was greater than in Crabapple for dogwood leaves (PR>F .0049) but not significantly different for beech (PR>F .9661). Beech leaves underwent much less total processing than dogwood leaves over the sampling period but all regression coefficients were significantly greater than zero. Dogwood leaves probably are palatable to macroinvertebrate shredders almost as soon as they enter the water while beech leaves require a long period of conditioning by microbes to become palatable. The faster processing

rate of dogwood leaves in Bruce Creek indicates that it may have a relatively larger macroinvertebrate shredder population. As will be seen in Chapter 3 this is, in fact, the case. The lack of any difference for beech leaves indicates there is no difference in the rate of microbial conditioning between the two streams or it has not had enough time to develop.

The greater density of shredders found in Bruce Creek may not be sufficient to account for the total difference in processing rate. Structural differences in Bruce Creek could conceivably effect processing rates. If a greater amount of CPOM in Bruce Creek was unavailable to shredders, perhaps due to sedimentation, this would cause the realized density of shedders in the leaf packs there to be greater than their absolute density would indicate. Another potential factor is abrasion resulting from greater sediment transport in Bruce Creek. Matter and Ney (1980) found greater amounts of FPOM in the substrate of surface mined streams (reclaimed) than in the substrate of an unmined stream, which they attributed to entrapment in the thick sediment. However, the organic fraction of the substrate of mined streams was less than in unmined stream. They concluded that much of the organic matter of the mined streams was probably unavailable to the stream macroinvertebrates. Unfortunately the present methodology cannot resolve processing rates into all of its components, nor can it determine what fraction of the POM resources of the stream bottom are actually available for processing. In the case of Bruce Creek, processing of some leaves definitely occurs at faster rates and therefore it is functionally differ-

ent. Part of this difference is no doubt due to a greater absolute density of shredder macroinvertebrates but other factors may contribute.

Webster and Waide (1982) found that processing of white oak and rhododendron leaves increased after clearcut logging but processing of dogwood leaves was unchanged. Processing of all three leaf species was reduced during logging. They felt that increased NO_3 concentrations after logging might have enhanced microbial conditioning, while increased flows after logging would have increased processing rates because of abrasion, and decreased leaf input to the stream after logging may have resulted in rapid ingestion by macroarthropods of the experimental leaves. They speculated that although total macroarthropod density was reduced after logging, so was the amount of POM, causing the leaf bags to attract greater relative density of shredders. Benfield and Webster (1985) reported greater processing rates of leaves (dogwood, red maple, and white oak) in second versus first order streams. Again they attributed increased processing rates in second order streams to a greater realized density of shredders by the leaf bags. They speculated that the first order streams probably had a surplus of CPOM because they are more retentive of POM (Bilby and Likens 1980). There was not a significant difference in the standing stock of POM in Crabapple and Bruce Creek so that differences in realized shredder densities are either due to absolute differences in shredder densities or some other factor.

Elwood et al. (1981) tested the effect of PO_4 enrichment on the processing rate of red oak leaves in Walker Branch Creek, Oak Ridge, Tennessee. They found that processing was significantly faster with

enrichment, but there was no difference between two enrichment levels. They concluded that most small streams are probably phosphorous, but not nitrogen, limited. The concentrations of NO_3 and PO_4 were not measured in the present study, but in order to evaluate the effects of surface mining on their concentrations in streams an analysis of variance was performed on NO_3, NH_3, PO_4 and total P values reported by Dyer (1982) for a large number of small streams in eastern Tennessee. The streams varied in mining history, including many unmined reference streams. Only streams without mapped buildings were compared. The null hypothesis tested was that mining history has no effect on these nutrient concentrations in streams. Streams were classed as unmined, mined before 1972 and mined after 1972. There were no significant differences between mining classes except for NH_3 in streams mined after 1972, probably reflecting fertilization to promote revegetation. It can only be said that Dyer's data were insufficient to reject the hypothesis that mining has no effect on nutrient concentrations. Many other factors that were not reported in the data could have caused elevated nutrient levels in the unmined streams. Among them are silvicultural, agricultural, and recreational uses of the unmined watersheds, but these factors would also be present in the mined watersheds.

The processing rates of dogwood from three independent studies in seven streams are presented in Table 2.7. Processing rates vary between streams and stream studies but there is usually a consistent species ranking within streams. Alder and dogwood are among the fastest, beech and rhododendron the slowest, and maples, hickories and oaks intermediate.

Table 2.7. The processing rate of dogwood leaves (day $^{-1}$) from three independent studies on seven streams.

Stream	Rate (± 95% CI)	Reference
Campbell Co., TN		Present study
Crabapple	0.0458 (± .0186)	
Bruce	0.0904 (± .0236)	
Coweeta, NC		Webster and Waide 1982
Big Hurricane Branch		
before logging	0.0219 (± .0025)	
during logging	0.0134 (± .0023)	
after logging	0.0219 (± .0024)	
Coweeta, NC		Benfield and Webster 1985
Stream 1	0.0486 (± .0159)	
Stream 2	0.0215 (± .0062)	
Stream 3	0.0102 (± .0051)	
Stream 4	0.0055 (± .0015)	

Conclusion

The main objective of this study was to assess the long term impacts, if any, of surface mining coal on the physical structure and biological function of small streams in the Cumberland Mountains of eastern Tennessee. The organic matter dynamics of two streams, one with approximately 3% of its surface area disturbed by mining 15 years prior to the study and the other a pristine reference, were compared as indicators of structural and functional changes. The differences in POM dynamics between the two streams are summarized in Table 2.8. The greater measured input of POM to Bruce Creek was not significantly

different except for POM input by blow-in across benches. Total standing stock of POM was also similar in the two streams. Transport of POM was not accurately measured by the drift nets employed in this study. The volatile portion of suspended solids, presumably very fine organic solids, was twice as great in Bruce Creek, and this was highly significant. The rate of leaf processing was significantly greater for flowering dogwood in Bruce Creek but was not significantly different for beech.

Table 2.8. Summary of observed differences in POM parameters between Crabapple and Bruce Creeks.

		Crabapple			Bruce		
Parameter	(units)	CPOM	FPOM	Wood	CPOM	FPOM	Wood
fall-in	(g/m^2)	374.9	22.1	65.2	388.1	22.1	80.3
blow-in	(g/m)						
bench		44.7	ns	5.4	119.8	ns	5.4
slope		265.4	ns	35.4	423.7	ns	58.2
total input	(g)	685.0	22.1	106.0	932.6	22.1	143.9
standing stock	(g/m^2)	27.8	4.4	13.4	18.1	2.3	15.0
transport							
drift net	(g/hr)	0.8	0.4	0.1	0.7	0.4	0.2
suspended solids (volatile)	(mg/l)	ns	2.7	ns	ns	5.1	ns
processing rate	(day^{-1})						
beech		-0.0022			-0.0021		
dogwood		-0.0458			-0.0904		

ns = not measured

Bruce Creek is a slightly smaller stream than Crabapple Creek and the retention of POM is usually greater in smaller streams (Bilby and

Likens 1980). It is somewhat surprising that, with similar input, Bruce Creek did not accumulate a larger standing stock of POM than Crabapple Creek. Bruce Creek may not retain POM as well as would be expected of a stream its size and/or it may process it faster. Both possibilities would imply that Bruce Creek is structurally and functionally different from Crabapple Creek and that long term effects resulting from surface mining coal are likely responsible. The faster processing of dogwood leaves in Bruce Creek also suggests a higher density of shredders in this stream. It will be seen in Chapter 3 that Bruce Creek does in fact have a 211% greater density of shredders. Whether they account for the increased processing rate of dogwood leaves depends on the relative density of shredders to CPOM. Since standing stock of CPOM is similar in the two streams it is likely that the relative density of shredders to CPOM is greater in Bruce Creek.

Dogwood leaves have a high nitrogen content rendering them palatable to most macroinvertebrate shredders soon after they they enter the water. They provided a good indicator of processing activity of the macroinvertebrate shredders. Beech leaves which need a long period of microbial conditioning would be a good indicator of the combined activities of the microbial and macroinvertebrate communities. The similar but small amount of processing of beech leaves in the two streams indicates that conditioning was similar but incomplete and macroinvertebrate shredding had not progressed far. A comprehensive description of taxonomic and functional group composition for these streams is presented in Chapter 2.

CHAPTER 3

TAXONOMIC AND FUNCTIONAL GROUP COMPOSITION OF MACROARTHROPOD COMMUNITIES IN A MINED VERSUS A PRISTINE STREAM IN THE CUMBERLAND MOUNTAINS OF EASTERN TENNESSEE

Introduction

The long-term effects of surface mining coal on the macroinvertebrate communities of small streams in eastern Tennessee were investigated by comparing the fauna of two streams differing in mining history. One stream, Crabapple Creek, was a pristine reference stream. The other, Bruce Creek, was mined without reclamation about 15 years prior to sampling. Approximately 3% of the surface area of Bruce Creek watershed has been disturbed by mining, a small amount relative to most stripmined watersheds in the region. The object of this study was to compare the density, number of species, indices of diversity, taxonomic and functional feeding group composition of the fauna of these two streams. A second objective of this study was to ascertain the relative value of structural versus functional descriptions of the macroarthropod communities of mined streams. The community structure of polluted streams including streams affected by surface mining of coal has been extensively studied. This literature provides some perspective for interpreting the diversity values for Crabapple and Bruce Creeks. Functional feeding group composition has been reported for a handfull of pollution studies including one surface mining study in southwestern Virginia (Matter 1978).

Evaluation of the effects of surface mining coal and other environmental disturbances affecting streams has in the past concentrated on measuring changes in the structure of biological communities. Commonly used measures of community structure include presence and absence and/or abundance of critical species (indicator organisms), species richness (number of species), relative or absolute population densities and diversity (a numerical expression of community composition based on its richness and evenness (relative abundances)). Diversity indices have been widely promoted (Wilhm and Dorris 1968, Cairns and Dickson 1971, Weber 1973). There has been some debate about the use of diversity indices in aquatic ecology (Green 1979, Kovalak 1981). Green (1979) argued that for the same effort required to compute a diversity index a direct statistical analysis would provide more information. He outlined a set of statistical strategies covering a wide range of environmental monitoring situations. Both approaches share the need for an accurate and consistent taxonomic foundation. Unfortunately, the taxonomy of lotic invertebrates is always unsettled and has been described by Cummins (1974) as "perpetually incomplete." He advocated a functional group analysis of aquatic biological communities as an alternate approach to the study lotic ecosystems which would partially avoid taxonomic problems and promote better understanding of lotic ecosystems. The trophic ecology of aquatic insects has been reviewed by Cummins (1974), Cummins and Klug (1979), and Merritt and Cummins (1984). Wiederholm (1984) provided a recent review of the responses of aquatic insect communities to a variety of environmental pollutants.

Criticisms of the use of diversity have some merit. There has probably been a tendency to settle on the use of diversity because it is easy to calculate and because it is so widely reported, when other analyses could have provided additional information. It is true that in all probability no two diversity indices rest upon the same taxonomic base or classification decisions. There are always some specimens too small or damaged to be assigned to species or even higher taxonomic categories. In addition, the accuracy of taxonomic classification is continually increasing as new taxonomic keys become available. Yet diversity indices consistently reflect detrimental impacts to stream communities in comparative studies. However, by reducing all of the faunal characteristics of a complex community, such as stream macroarthropods, to an index, a great deal of information is lost. More insight into the ultimate causal interactions between the physical and biological components of the stream ecosystem can be gained by expanding the study into community function. This study pursued both approaches allowing a comparison of the relative merit of each.

Several studies in eastern Tennessee have documented the effects of surface mining coal on the community structure of streams (Minear et al. 1976, 1977, 1978, 1979, 1980, Talak 1977, Tolbert 1978, Vaughan 1979, 1980, Vaughan et al. 1978, 1982). Upon the onset of mining the abundances and diversities of diatom, fish, and macroinvertebrate communities are drastically reduced. Approximately five years after mining ceases, abundance and diversity of fishes and macroinvertebrates begin to recover, reaching pre-mining levels within 20 years. Diatoms are most reduced and take much longer to recover than fish and macroinver-

tebrates (Vaughan et al. 1978, Vaughan 1979, 1980). Although quantitative indices of community structure such as density and diversity indices recover within 20 years, differences in the taxonomic composition between mined and unmined streams may persist (Talak 1977, Vaughan 1978, 1979, Vaughan et al. 1978, 1982, Williams 1981). Long term changes in the taxonomic composition of mined streams suggest that other means of describing the environmental impact of surface mining coal should be explored. In addition, advances in the theory of lotic ecology and development of functional models of stream ecosystems provide a number of testable hypotheses with which to direct research concerning the impacts of surface mining coal.

Materials and Methods

Physical and chemical water properties were measured as described in Chapter 2. Aquatic macroinvertebrates were sampled approximately monthly with eight standard Surber samples (0.1 m^2 each) per stream. Samples were obtained on six dates between 7 January and 26 May 1984. Determining an adequate number of benthic samples is a problem which has recieved considerable attention (Gaufin et al. 1956, Needham and Usinger 1956, Chutter and Noble 1966, Resh 1979). Estimates of the adequate number of Surber samples to achieve satisfactory precision range from 3 (Chutter and Noble 1966) to 76 (Needham and Usinger 1956). Resh (1979) pointed out that life history features of the organisms being sampled are just as significant a source of variation as mechanical problems with the sampling equipment. All studies generally agree that sample stratification and careful selection of the sampling universe can

greatly improve the precision of sampling. Resh (1979) pointed out, however, that most published benthic studies are based on too few samples to be statistically valid. Tolbert (1978) determined that eight paired square foot Surber samples achieved a sampling precision of 10% or less. In this study 8 single square foot samples were collected for several reasons. First, it was felt the distribution and number of samples were more important than the total area sampled. Second, paired samples only increase the area sampled and not the spatial distribution of samples, and can even limit the spatial distribution because fewer areas of the streambed will accommodate paired samples. Difficulty in placing the larger paired samples in the streams might actually result in lower capture efficiencies and greater sampling variation. Finally, sample processing time is proportional to the total area sampled but precision is proportional to the number of samples. Both streams were sampled on the same day to minimize potential differences in sampling efficiency due to meteorological or seasonal effects. The 95% confidence intervals for productivity estimates described in Chapter 4 generally range between 20 and 50% of the estimate but were adequate to resolve significant differences in productivity for most of the fauna studied. This result was taken as evidence that the sample size in this study was indeed adequate.

Sample locations were chosen randomly along the stream course with sampling proceeding in the upstream direction. Riffle areas were the only habitat sampled. The sampler was placed on the streambed as firmly as possible. Any rocks or other obstructions which interfered with placing the sampler secure to the bottom were removed and processed

first in order to improve the placement of the sampler. Larger rocks which partly overlapped the sampling area were cleaned of macroarthropods from the area of overlap only. There was an element of subjectivity involved here but this approach does avoided the subjectivity of placing the sampler only in locations without such rocks. Leaves and other organic matter, even when encountered in abundance, were preserved along with the macroarthropods in 95% ethanol and stored in 250 ml Nalgene sample cups. Some researchers discard this material, but many organisms can be lost this way and it is also an important parameter in and of itself.

Detritus (POM) and macroinvertebrates were separated from sand and gravel by sucrose flotation (specific gravity 1.12) (Pask and Costa 1971). The sample was stirred in approximately 1000 ml of the sucrose solution and the macroarthropods and detritus (particulate organic matter [POM]) floating in the supernatant were decanted through a US Standard series 120 mesh sieve. After repeating this process three times, the residue of sand and gravel was spread on a white porcelain tray and checked for missed specimens. Usually only a few stone cased trichopterans and chironomids were not separated by the floatation. The residue of all samples was checked to insure that all the trichopterans (mostly Neophylax spp.) were included in the analysis. The POM and macroarthropods were then rinsed under a gentle stream of tap water and stored in 95% ethanol until being separated by hand. The macroarthropods were subsequently separated from POM by hand picking in a white porcelain tray and stored in 80% ethanol for later identification.

Macroarthropods were defined as larval aquatic insects (exclusive of Chironomidae and Ceratopogonidae) and the aquatic isopod, Lirceus fontinalis, for the purposes of this study. Other Crustacea besides L. fontinalis, especially crayfish, were collected in the Surber net but this method of sampling was very inefficient for them. Some aquatic insect species have adult forms which live in streams but are capable of migrating between streams. The adult forms of these species were excluded from the analysis whereas the larvae of these species were included as they must survive and complete their development in one stream. Chironomids and ceratopogonids were excluded because of the tremendous increase in time required to adequately hand pick all of them from the mineral and organic residue of the samples and because Surber sampling was not very efficient for these organisms. Exclusion of chironomids and ceratopogonids was not intended to diminish the importance of these abundant and diverse dipterans in the overall ecology of the stream.

Macroarthropods were identified to species when possible. However, a few troublesome groups were difficult to identify even to genus. Keys used in identification were: general, Brigham et al. (1982); Coleoptera, Brigham (1982); Diptera, Webb and Brigham (1982); Ephemeroptera, Allen and Edmunds (1963, 1965), Edmunds et al. (1979), Bednarik and MacCafferty (1979) Unzicker and Carlson (1982); Isopoda (Lirceus), Hubricht (1949); Lepidoptera, Brigham and Herlong (1982); Megaloptera, Brigham (1982); Odanata, Huggins and Brigham (1982); Plecoptera, Frison (1942), Hitchcock (1974), Unzicker and McCaskill (1982); Trichoptera, Schuster and Etnier (1978), Wiggins (1978), Unzicker et al. (1982). Many species

could be reliably identified as small as 2 mm or less. This was important for the estimates of secondary productivity discussed in Chapter 4.

Species diversity and other indices of community composition were based on all unique taxonomic groups which could be reliably distinguished. Individuals were always assigned to the lowest taxonomic level achievable but not all of the taxonomic classifications were exclusive so the taxonomic checklist was slightly larger than the number of taxa included in diversity calculations. Shannon diversity (Shannon and Weaver 1949), dominance (Simpson index) (Simpson 1949), and evenness (Pielou 1975) were calculated as follows:

Shannon diversity: $H' = - \Sigma P_i \log P_i$

Simpson index: $\lambda = - \log \Sigma P_i^2$

Eveness: $J' = H' / \log S$

where: P_i equals the proportion of total individuals in species i, and

S equals the total number of species (estimated from the total number of species found in each stream over all samples),

using a SAS program (SAS Institute Inc. 1982). Pielou (1975) provided a comprehensive review of the indices presented here and a detailed discussion of their underlying assumptions and use. There has been considerable debate concerning the appropriate use of particular diversity indices. The Shannon Index, H', presented here is not necessarily the best index but the most often reported. The Shannon index assumes an infinitely large population is sampled; however, in reality only a sample of the population is actually measured. Pielou considers that the Brillouin Index (Brillouin 1962) is the most appropriate diversity index to apply to samples because it is free from sampling error.

Considering the imprecision of aquatic insect taxonomy and the sampling error involved in collecting aquatic macroinvertebrates this point is probably trivial.

Results and Discussion

Water quality data for Crabapple and Bruce Creeks are presented in Appendix A, Table A1. The differences in the physical and chemical properties of the water are summarized in Chapter 2.

The effects of surface mining coal on water quality of small streams include increased sedimentation, altered hydrology (increased peak flows), and rarely acidification. Acidification usually occurs in association with underground mining (Boccardy and Spaulding 1968) and stripmining confounded by auger operations and post extraction processing facilities. Auger mining is no longer practiced in eastern Tennessee and most surface mine drainage in this region is alkaline. Acidification is a serious problem in many areas throughout the eastern coal province, but most of the studies referred to in this dissertation will not involve acid mine effects unless explicitly indicated. One such study by Herricks and Cairns (1974) demonstrated the effects of acid mine drainage on natural stream sections in Pennsylvania and a stream section in Virginia which they experimentally acidified. They observed reductions in abundance and diversity of macroinvertebrates as a result of acidification that were similar to those observed for non-acid mine situations. The effects of acid mine drainage on aquatic insects was reviewed by Roback and Richardson (1969).

Tolbert (1978) concluded that sedimentation and increased peak flows were the main factors associated with the reduction in abundance and diversity of macroarthropod communities in small streams in eastern Tennessee. Stair and Tolbert (1980) conducted a velocity tolerance study for several species of insects found in eastern Tennessee streams. They concluded that the increased flow velocities resulting from surface mining, over two times greater during storm events, could be a important factor in the reduced abundance of many macroarthropod species in mined streams.

A complete taxonomic checklist for the entire sampling period, including absolute and percent abundance of each taxon, is presented in Table 3.1. A total of 86 taxa was distinguished from 13,776 individuals collected from both streams. This list includes all taxonomic determinations that were made even if all individuals in the next taxonomic level could not be assigned to one of these lower levels. For example, three species of Stenonema mayflies were identified but not all Stenonema mayflies could be assigned to one of these three species. Similarly, five genera of tipulids were identified but not all individuals could be assigned to one of these genera.

The taxonomy of aquatic insects is constantly changing and rapidly improving. It is important for authors to clearly and implicitly define the classification decisions which they have inevitably had to make regarding individuals belonging to groups for which taxonomy is incomplete or which are too small or damaged to be completely identified. This is especially true for studies where the number and relative abundance of taxa can significantly effect the value of crucial parameters

Table 3.1. Taxonomic checklist, including total and % abundance of each taxon, based on 48 Surber samples from each stream over all six sample dates from January to May 1984.

Taxa	Crabapple Creek Total	%	Bruce Creek Total	%
Coleoptera				
Elmidae				
Ukn	168	2.11	26	0.45
Eubriidae				
Ectopria nervosa	276	3.46	50	0.86
Hydrophilidae				
Hydrophilid sp.	1	0.01	0	0.00
Psephenidae				
Psephenus herricki	272	3.41	41	0.71
Ptilodactyidae				
Anchytarsus bicolor	0	0.00	1	0.02
Ukn				
Ukn	1	0.02	0	0.00
Diptera				
Simuliidae				
Ukn	283	3.55	115	1.99
Tipulidae				
Antocha sp.	4	0.05	1	0.02
Dicranota sp.	0	0.00	4	0.07
Erioptera sp.	3	0.04	3	0.05
Hexatoma spp.	378	4.74	72	1.24
Tipula sp.	11	0.14	4	0.07
Ukn	5	0.06	2	0.03
Ukn				
Ukn	11	0.14	3	0.04
Ephemeroptera				
Baetidae				
Baetis cf. brunneicolor	448	5.62	203	3.51
Ephemerellidae				
Drunella cf. cornutella	21	0.26	3	0.05
Ephemerella dorothea	1769	22.18	547	9.46
Eurylophella spp.	5	0.06	0	0.00
Ukn	3	0.04	0	0.00
Ephemeridae				
Ephemera guttulata	11	0.14	23	0.40

Table 3.1. (continued).

Taxa	Crabapple Creek Total	%	Bruce Creek Total	%
Heptageniidae				
Cinygmula subaequalis	355	4.45	19	0.33
Epeorus dispar	529	6.63	683	11.81
Heptagenia sp.	20	0.25	3	0.05
H. thetis	8	0.10	1	0.02
Stenacron carolina	1	0.01	0	0.00
S. interpunctatum	3	0.04	0	0.00
S. pallidum	2	0.03	4	0.07
S. sp.	2	0.03	2	0.03
Stenonema carlsoni	1	0.01	0	0.00
S. meririvulanum	5	0.06	0	0.00
S. modestum	0	0.00	1	0.02
S. spp.	1	0.01	1	0.02
S. vicarium	1	0.01	0	0.00
Ukn	0	0.00	2	0.03
Leptophlebiidae				
Habrophlebia vibrans	8	0.10	0	0.00
Habrophlebiodes americana	788	9.88	57	0.99
Paraleptophlebia guttata	12	0.15	16	0.28
P. moerens	31	0.39	0	0.00
Ukn	3	0.04	9	0.16
Siphlonuridae				
Ameletus lineatus	459	5.76	213	3.68
Isopoda				
Asellidae				
Lirceus fontinalis	136	1.71	1559	26.95
Lepidoptera				
Ukn				
Ukn	5	0.06	2	0.03
Megaloptera				
Corydalidae				
Nigronia fasciatus	4	0.05	4	0.07
Sialidae				
Sialis americana	1	0.01	1	0.02
Odonata				
Coenagrionidae/Libullidae				
Ukn	0	0.00	2	0.03

Table 3.1. (continued).

Taxa	Crabapple Creek Total	%	Bruce Creek Total	%
Cordulegastridae				
Cordulegaster sp.	9	0.11	0	0.00
Gomphidae				
Lanthus sp.	9	0.11	4	0.07
Ukn	1	0.01	1	0.02
Ukn				
Ukn	1	0.01	1	0.02
Plecoptera				
Capniidae				
Allocapnia sp.	4	0.05	8	0.14
Paracapnia opis	57	0.71	111	1.92
Ukn	1	0.01	0	0.00
Chloroperlidae				
Hastaperla brevis	183	2.29	95	1.64
Sweltsa/Suwallia spp.	29	0.36	53	0.92
Ukn	25	0.31	8	0.14
Leuctridae				
Leuctra truncata	52	0.65	99	1.71
Nemouridae				
Amphinemura delosa	169	2.12	770	13.31
A. delosa/wui	17	0.21	19	0.33
A. wui	265	3.32	118	2.04
Ukn	3	0.04	12	0.21
Peltoperlidae				
Peltoperla cf. maria	129	1.62	179	3.09
Perlidae				
Acroneuria carolinensis	62	0.78	70	1.21
Perlodidae				
Isoperla holochlora?	64	0.80	30	0.52
Remenus bilobatus	1	0.01	0	0.00
Ukn	11	0.14	3	0.05
Ukn sp. a	40	0.50	28	0.48
Ukn sp. b	1	0.01	4	0.07
Ukn sp. d	1	0.01	0	0.00
Yugus/Malirekus spp.	29	0.36	27	0.47
Perlidae/Perlodidae				
Ukn	13	0.16	1	0.02

Table 3.1. (continued).

Taxa	Crabapple Creek Total	%	Bruce Creek Total	%
Trichoptera				
Hydropsychidae				
Cheumatopsyche sp.	1	0.01	1	0.02
Diplectrona modesta	91	1.14	222	3.84
Hydropsyche sp.	7	0.09	25	0.43
Lepidostomatidae				
Lepidostoma sp.	73	0.92	9	0.16
Limnephilidae				
Goera cf. calcarata	20	0.25	7	0.12
Neophylax cf. mitchelli	97	1.22	44	0.76
N. sp.	336	4.21	101	1.75
Pycnopsyche sp.	6	0.08	9	0.16
Philopotamidae				
Wormaldia sp.	69	0.87	6	0.10
Polycentropodidae				
Polycentropus sp.	42	0.53	23	0.40
Rhyacophilidae				
Rhyacophila carolina	0	0.00	1	0.02
R. glaberrima	0	0.00	1	0.02
R. ledra	3	0.04	4	0.07
R. cf. mycta	4	0.05	2	0.03
R. cf. nigrita	4	0.05	4	0.07
R. spp.	13	0.16	11	0.19
total	7987		5788	

in the analysis such as diversity indices. The basic strategy employed in this study was to assign all individuals to the lowest possible taxonomic level. Then the reliability of the taxonomic designation for all individuals in that group was reassessed prior to calculating diversity and some reaggregation or exclusion of taxa was occasionally necessary. A brief but comprehensive review of the faunal list will clarify this point.

The Coleoptera represent one of the most straightforward taxonomic groups because three of the five families contain only one species each. The remaining two families were easily separable from each other and the other families, although both were probably represented by more than one species (perhaps even genera) in the streams. Adult beetles were collected and saved but were not included in this study. Dryopidae (Helichus) adults were also collected but are not included because only larvae were under consideration. The one coleopteran which was not identified to family was included in the diversity calculations because it was not one of the previously identified families.

The Diptera have a very large and diverse aquatic fauna. The Chironomidae and Ceratapogonidae were not included in the above faunal list because there was not enough time to sample them quantitatively. Some individuals were retained from each sample in order to present a faunal list of the major taxa present in these two families for each stream (Appendix B, Table B1 (Pennington 1985)). The Tipulidae are the largest family of aquatic Diptera but keys for larvae do not separate specimens beyond genus. Hexatoma followed by Tipula were the most prominent genera of tipulids in the streams. The genus Hexatoma definitely

contained more than one species but they could not be consistently separated and so were lumped. The other genera were only occasionally sampled and some of these individuals may be semiaquatic. Tipulids which could not be assigned to genus were excluded from diversity calculations. The Simuliidae were represented by three genera, Cnephia / Simulium, Prosimulium, and Simulium. There were probably more than three species included in these three genera. The most prominant genera for the duration of the study was Prosimulium. An occasional dipteran not belonging to one of the above families was encountered and was assigned to the taxonomic category "ukn" family. Each of these individuals was included in the diversity calculations as one taxon because they definitely did not belong to one of the above families.

Ephemeroptera are usually the dominant order in small undisturbed streams in this area. They are a large and diverse order but the taxonomy for most groups was relatively good. The Baetidae are, however, a difficult group. There seemed to be only one species in the two streams which has tentatively been designated B. brunneicolor. The Ephemerellidae were reliably identified to species except for the Eurylophella, which includes but may not be limited to E. funeralis. The most numerous heptageniids, Epeorus and Cinygmula, could be reliably distinguished for all specimens and contain a single species each. The remaining Heptageniidae: Heptagenia, Stenacron, and Stenonema, were difficult to take to species consistently. Each genus was probably represented by at least two to three species in the stream. The taxonomy of the Stenonema mayflies is especially frustrating and will undoubtedly continue to change. The specimens which could not be assigned to species or genus

were not included in the diversity calculations. Almost all the Leptophlebiidae could be distinquished reliably on the basis of characters which were not in standard keys and on overall morphology. The primary character used to separate genera in the Leptophlebiidae was gill morphology. However, leptophlebiid nymphs often lose all of their gills in storage. It was possible to separate Habrophlebia from the Paraleptophlebia on the basis of fore femoral armature and Habrophlebiodes from Paraleptophlebia by the notched labium. Paraleptophlebia moerens differed from P. guttata in tarsal denticles and general leg morphology. The few individuals which could not be assigned to genus were not included in diversity calculations as they probably belong to one of the identified groups.

The isopod Lirceus fontinalis was identified by an old key (Hubricht 1949); this genus may soon undergo taxonomic revision. In all probability this would result in no more than a name change for what appears to be the only species in the stream.

Three orders (Lepidoptera, Megaloptera, Odonata) occured at very low frequencies in the streams studied here. Only seven lepidopterans were collected and were not identified to family. The few Megaloptera were easily identified to species, but all the Odanata were very small and some could not be identified to family. However, at least four distinct taxa were believed to be present and were included in the diversity calculations.

The Plecoptera contained several difficult taxonomic groups but most genera could be distinguished. The Chloroperlidae were problematical because young Hastaperla brevis and Sweltsa / Suwallia specimens

were indistinguishable. The composite generic taxa, Sweltsa / Suwallia definitely included Sweltsa mediana but at least one additional species was suspected. The Chloroperlidae which could not be assigned to one of these two taxa were not included in the diversity calculations. The Nemouridae were dominated by two species, Amphinemura delosa and A. wui, which could be reliably distinguished except for very small or damaged individuals. At least one other genus was occasionally collected but could not be identified. Three taxa of Nemouridae were included in the diversity indices. The perlodid stoneflies were a very difficult group but at least seven taxa in at least three genera could be distinguished on the basis of mouthparts. The Yugus / Malirekus taxon may contain both genera. After viewing a large series of specimens it was obvious that a complete spectrum of intermediate color morphs occured in these taxa, and they cannot be reliably separated. Specimens typical of both genera were collected in the same Surber sample. The lacinia of these genera was a distinctive and reliable character even for very small (< 2 mm) individuals. Occasionally an extremely small (<1 mm) nymph could not be assigned to genus or family, and these were excluded from the diversity calculations.

The Trichoptera were reliably separated to genus and most to species. The genus Neophylax is under revision but a distinct species keyed out as N. cf. mitchelli on the basis of its prominant frontoclypeal tubercle. The distribution N. mitchelli is probably limited to the Blue ridge physiographic province. The remaining specimens in this genus may contain more than one species. The genus Rhyacophila contained five taxa whose characters agreed well with those of nominal

species known to occur in the area (Etnier and Shuster 1979). Approximately half of all Rhyacophila individuals could not be assigned to species but almost certainly belong to one of the five previously identified and were not included in the diversity calculations.

A total of 7987 individuals was collected from Crabapple Creek and 5788 individuals from Bruce Creek, a difference of 28%. Table 3.2 compares the number of species (N), total number of individuals (I), Shannon diversity (H'), dominance (Simpson index (λ)), and evenness (J') for Crabapple and Bruce Creeks based on eight pooled Surber samples for six dates. Crabapple Creek had consistently more total species (richness) and individuals (density) in all months and greater Shannon diversity, evenness in the distribution of individuals between taxa, and dominance indices in all months except for early May. Crabapple Creek had greater mean values for all five parameters than Bruce Creek. Lower abundance and diversity is typical of stripmined streams and streams affected by other kinds of environmental disturbances. However, the difference in diversity between Crabapple and Bruce Creeks was not great. In fact Bruce Creek would probably compare favorably to some unmined streams in diversity.

Talak (1977) and Vaughan et al. (1978) surveyed 23 streams in the coal mining areas of east Tennessee which varied in time since mining including three unmined reference streams. They found that a chronology of recovery could be described in terms of total density, species richness, and diversity of aquatic insects and fish. Most streams experienced drastic reductions in diversity, richness, and density following mining, reaching a minimum in about four years followed by a gradual

Table 3.2. Number of species, N, total number of individuals, I, Shannon diversity, H', Simpson index, λ, and evenness, J', for eight Surber samples each from two streams on six sample dates from January to May 1984.

	Crabapple					Bruce				
Date	N	I	H'	λ	J'	N	I	H'	λ	J'
10784	44	1574	2.57	3.78	0.679	35	1223	2.21	3.56	0.622
20484	49	1846	2.68	3.89	0.689	39	1209	2.23	3.66	0.607
30384	43	1448	2.92	3.76	0.776	39	1006	2.56	3.66	0.699
33184	41	1426	2.76	3.71	0.742	39	1011	2.53	3.66	0.690
50584	39	885	2.73	3.66	0.746	37	644	2.75	3.61	0.762
52684	37	697	2.64	3.61	0.731	34	635	2.45	3.53	0.695
mean	42	1313	2.72	3.73	0.727	37	955	2.45	3.61	0.679

increase to approximate pre-mining values within 20 years. Four of the streams Talak studied had been mined 15 years prior to sampling, as is the case for Bruce Creek in this study. Talak found reduced abundance and diversity in these four streams as compared to references streams. Williams (1981) studied two streams mined over 20 years previously and two reference streams. She found a greater density and species richness for the reference streams but no significant differences in diversity between the reference and mined streams. Matter and Ney (1981) found greater densities, biomass, and diversity of the benthic macroinvertebrate community in the reference stream that they compared to two reclaimed stripmine streams in southwest Virginia. They noted that other streams which had not been reclaimed had similar densities to

those which had been reclaimed. However, the mined streams in this study had been recovering for only a few years, the low point in the water quality and biological recovery of mined streams (Talak 1977, Vaughan et al. 1978, 1982).

Herricks and Cairns (1974) observed reduced diversity and density of macroinvertebrates in an experimental section of stream in Montgomery Co., Virginia, which they artificially acidified. They also noted higher densities and diversities of macroinvertebrates in the unpolluted reference sections of a stream system affected by acid mine drainage in Fayette Co., Pennsylvania. Specht et al. (1984) found that density, number of species, and diversity of benthic invertebrates decreased downstream from where the effluent of a coal fired power plant settling pond entered the stream. In both stream studies where only a small section of stream was affected, the communities recovered quickly after the source of pollution was abated. This was because a healthy, intact macroinvertebrate community upstream provided a rapid colonizing source and clean upstream water quickly purged the affected stream sections of residual pollution. The situation in surface mined watersheds is just the opposite. Disturbance usually occurs at the very source of the streams and is only gradually dissipated downstream. The disturbance is long term and chronic and only gradually diminishes with time. While recovery of biological communities from most sources of disturbance can be measured on a biological time scale of weeks to perhaps years, it is possible that recovery from surface mining must be considered on a geological time scale of at least decades if not centuries. In addition to the absolute difference in density of macroarthropods, there were

many differences between the percent abundances of individuals within major taxonomic levels. Figure 3.1 compares the percent composition of each stream contributed by the nine major orders. Crabapple Creek had four times as many coleopteran individuals, over twice as many dipterans, and almost twice as many ephemeropterans as Bruce Creek as a percent of the total fauna. Ephemeroptera remained the dominant order in both streams but in Bruce Creek the Plecoptera and Isopoda were a close second and third, respectively, in percent abundance. Bruce Creek had more than 10 times as many Lirceus fontinalis, and twice as many plecopterans as a Crabapple Creek in percentage terms. The two streams were comparable in the percent composition of trichopterans. The orders Lepidoptera, Megaloptera, and Odonata were too poorly represented for comparative purposes.

Several studies have noted differences in taxonomic composition between mined and unmined streams (Talak 1977, Williams 1981, Matter and Ney 1981). Other environmental impacts such as the effluent of coal fired power plant settling basins also result in changes in the taxonomic composition of macroarthropod communities (Cherry et al. 1979, Specht et al. 1984). Ephemeroptera, which are usually the numerically dominant order in small streams throughout the eastern coal province, are often disproportionately reduced relative to various other taxa which may show a relative increase as percent of the total fauna. This is the exact opposite response to that reported for clearcut logging. Newbold et al. (1981) observed increased abundance of Ephemeroptera, Oligochaeta, and Chironomidae, but reduced abundance of Plecoptera as a response to clearcut logging. Different relative abundances of families

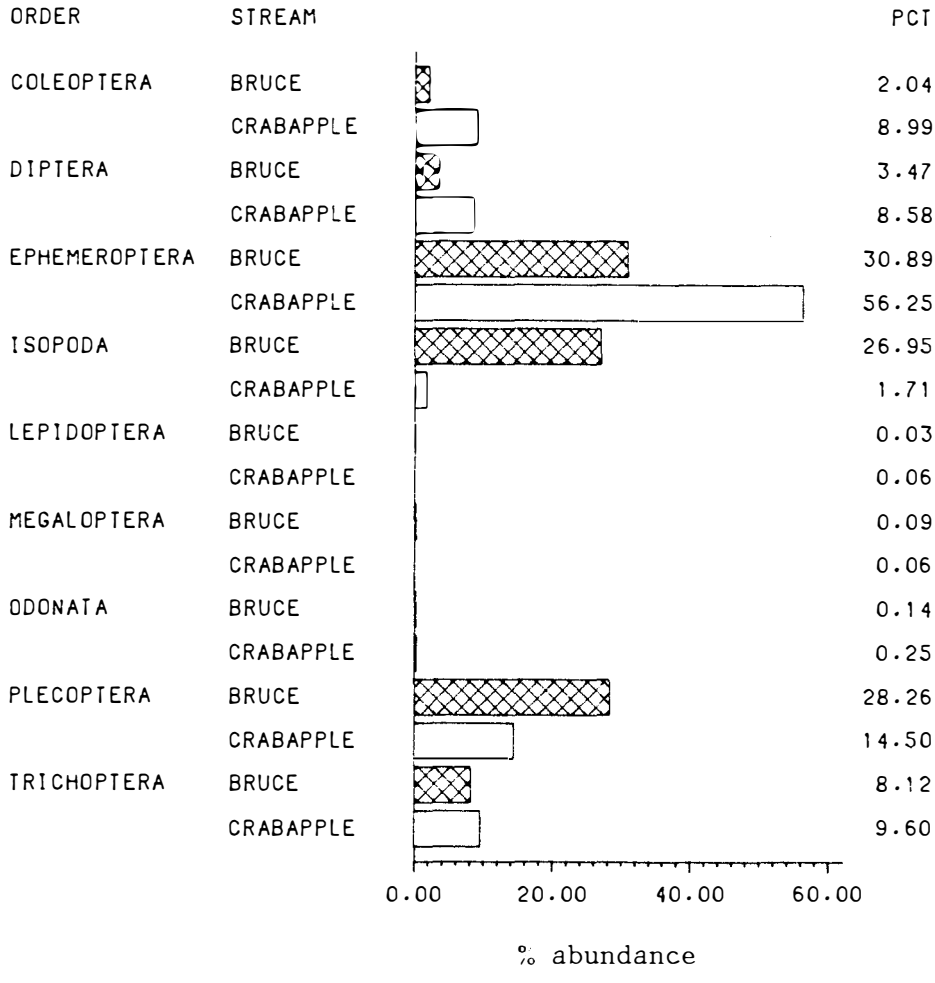


Figure 3.1. Percent of total number of individuals contributed by nine major orders of macroarthropods in Crabapple and Bruce Creeks.

within orders have also been observed in mined streams (Williams 1981). The selective differences in abundance of various families suggested that ecologically similar groups of species, not particular orders of macroinvertebrates, are adversely effected by surface mining.

Several studies investigating the effects of surface mining coal on the aquatic insect communities of the New River, Tennessee, have been conducted under the supervision of Dr. Gerald Vaughan, University of Tennessee, Knoxville. The first by Talak (1977) compared the macroarthropod composition of several small streams similar to Crabapple and Bruce Creeks in east Tennessee. These streams varied in the amount of time that they had to recover since mining. He found a smaller percentage of insects in the Ephemeroptera in mined streams compared to three unmined reference streams. The difference was greatest for streams mined 6-10 years previously and decreased thereafter. However, the 43% of individuals in the Ephemeroptera in the three reference streams seems a little low compared to other unmined streams in this region. In fact, the 51% of individuals in the Ephemeroptera in the streams mined over 20 years previously is probably still less than comparable unmined streams. Fifty-six percent of the individuals in Crabapple Creek were in the Ephemeroptera versus 27% in Bruce Creek. A disproportionate reduction in abundance was also was observed for Coleoptera, but Diptera, Plecoptera, and Trichoptera increased in relative abundance after mining before eventually returning to pre-mining levels in about 20 years. The relative abundance of the orders in his study might be different from these data because he sampled in June and July. Much of the fauna in this study would have matured and emerged by the time he sampled in June.

Tolbert (1978) followed up on this initial study by sampling four of the streams studied by Talak 10 times over the course of a year. Some of the differences in faunal composition between the reference and mined streams she studied were similar to differences observed in the present study. Ephemeroptera were only 11 and 16% of the fauna in two of the mined streams but 66% of the fauna in another of the mined streams. The reference stream in this study, Lowe Creek, had 52% of its fauna in the Ephemeropera. She found that all orders except Diptera were reduced in relative abundance in the mined streams, and all orders were reduced in absolute abundance in the mined streams. Heptageniids, ephemerellids, leptophlebiids, and leuctrids were among the families which were very reduced or absent from the mined streams. Amphinemura (probably delosa and/or wui) were not different in relative abundance in the four streams.

Williams (1981) investigated the long term effects of surface mining on aquatic macroinvertebrate communities. She sampled three of the same streams as Talak including Lowe Creek, the reference stream used by Tolbert, but she did not find as strong a relative contrast as Talak in the percentage composition between orders of the mined versus reference streams. However, the mined streams in her study had been recovering for over 20 years, longer than when Talak sampled them. She did find reduced numbers of ephemeropterans but they remained the dominant order in all four streams. She estimated that Ephemeroptera comprised 61% of the fauna of Lowe Creek. Thus the three independent estimates of the contribution of Ephemeroptera to the fauna of one stream, Lowe Creek, range from 52 to 72%. She sampled the same months

as this study. Williams's results may have been influenced by the choice of Indian Creek as one of the reference streams in her study. Indian Creek is a second order stream and can be expected to exhibit some faunal differences on this basis alone. On the other hand, the differences in percent composition of orders observed in these data may be magnified by the fact that Bruce Creek is a slightly smaller stream than Crabapple Creek. Faunistically, Bruce Creek seems to closely resemble Duncan Creek, studied by Williams(1981), especially with regard to the significant proportion of Isopods in both streams. These streams are both small and have a steep average gradient.

All of the above studies could differ from this study because of differences in sampling procedure. Talak collected qualitative samples with screens. Tolbert and Williams sampled twice the total stream bottom area as in this study, but the distribution of the eight samples was probably similar. A conscious effort was made not to avoid large rocks when placing the sampler on the stream bed and included any leaves and wood collected in the sample. The inclusion of large rocks and organic debris could significantly effect the relative abundance of certain taxa in the sample. For example, the coleopterans and limnephilid trichopterans might have been taken more frequently in samples than in the above studies. The other studies followed the same protocol; however, subtle differences in sample collection between individuals was still possible. Since these three studies were part of a continuing research effort, they should display more continuity with each other than with other studies reported in the literature. The comparisons of abundances within studies are certainly more valid than comparisons

between studies. Local factors can also be important sources of variation. There was also a considerable amount of variation in the extent of mining from stream to stream and the way it was mined. Even before reclamation laws were passed some operators were much more considerate of the land than others. The position and depth of cut along with local geology are yet other sources of variation contributing to stream by stream faunal differences.

Matter and Ney (1981) compared two stripmined streams which had received surface reclamation, three unreclaimed streams, and a reference stream in southwest Virginia. One of the unreclaimed streams was affected by acid mine drainage and will not be considered here. Ehemeroptera comprised 27% of the benthic invertebrate fauna in the reference stream and varied from 8 to 54% in the reclaimed streams and 4 to 29% in the unreclaimed streams. Diptera were 20% of the reference stream macroinvertebrate fauna and 19 to 47% of the reclaimed stream fauna and 46 to 70% of the unreclaimed stream fauna. Coleoptera were the only taxa to show consistent reductions in the mined streams relative to the reference stream. Coleoptera comprised 23% of the fauna in the unmined stream but not more than 3% of any of the mined streams. Matter and Ney (1981) employed a different sampling technique than in this or any of the other studies cited. Their reference stream was also larger than the two reclaimed streams, with a drainage area of 400 ha versus 120 to 130 ha, which again can be expected to introduce a source of variation to the taxonomic composition of the macroinvertebrate community.

Specht et al. (1984) found a different pattern of taxonomic change in a section of second order stream affected by effluent from a fly ash

settling pond for a coal fired power plant. Ephemeroptera, Plecoptera, and Trichoptera were reduced in abundance and the more tolerant Coleoptera (mostly Stenelmis spp. and Psphenus herricki) became the dominant taxa. After the effluent was eliminated the community composition recovered in a period of months. The resilience of the beetle community in this study was interesting since beetles were one of the orders with the greatest percent difference between mined and reference streams in the study by Matter and Ney. Coleoptera also showed considerable difference in relative abundance in the present study. Talak and Williams did not observe much difference in the relative abundance of Coleoptera between streams mined over 20 years previous to their survey and unmined streams. The Coleoptera, especially Psephenus and Ectopria, may depend on a healthy periphyton community. The diatoms of surface mined streams were the most drastically reduced and slowest to recover of all the stream biological communities (Vaughan et al. 1978, Vaughan 1979,1980). This may reflect water quality degradation which persists even after 20 years (Talak 1977, Vaughan 1979, Vaughan et al. 1978,1982).

While a pattern of percent change in community composition between orders can be inferred from these studies, it was not very consistent. However orders of aquatic insects contain a lot of ecological overlap. For example Coleoptera, Diptera, Ephemeroptera, Plecoptera, and Trichoptera all contain taxa which are predators, detrivores, and plant consumers and all contain taxa which overlap in the way they obtain food and in general habitat preferences.

A better view of faunistic differences between the mined versus reference streams might be gained by comparing the percent composition by family. At this level of resolution important ecological differences can be associated with the differences between families within orders. Figure 3.2 presents the percent composition by family of Crabapple and Bruce Creeks.

The eubriid and psephenid coleopterans were also less abundant in Bruce Creek relative to Crabapple Creek. Tipulids were relatively more abundant in Crabapple Creek also. Four of the Ephemeroptera families were less abundant in Bruce Creek relative to Crabapple Creek. These four families can all be categorized as collector-gatherers except for the Baetidae, some of which are periphyton grazers (Merritt and Cummins 1984). The Heptageniidae were the only Ephemeropteran family with similar abundance in Bruce Creek. An ephemerellid, Ephemerella dorothea, at 22% of the total fauna was the numerically dominant species in Crabapple Creek over the period of sampling. The isopod, Lirceus fontinalis (family Asellidae), comprising 27% of the fauna, was the dominant macroarthropod in Bruce Creek. The occurence of Lirceus fontinalis in the small streams of this area varies. While they are found in most small streams in low densities, some streams support large densities. Both Bruce Creek and Duncan Creek, studied by Williams (1981) contained large numbers. These streams have large gradients and probably do not contain many, if any, fish. Crabapple contained fish over about half its length while Bruce Creek did not seem to contain any although a rigorous search was never made for them. Streams with large gradients can be expected to dry up more frequently and contain more waterfalls,

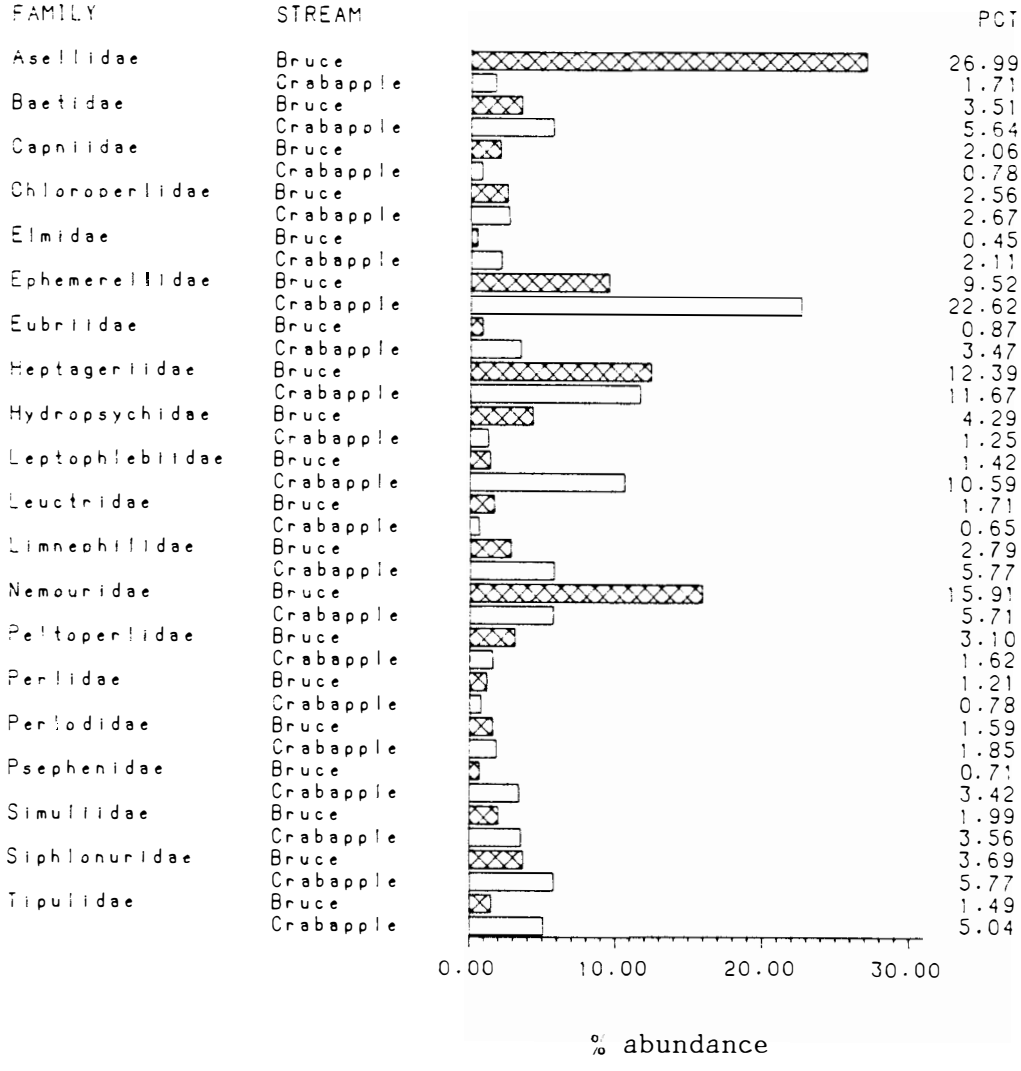


Figure 3.2. Percent of total number of individuals contributed by 20 families of macroarthropods in Crabapple and Bruce Creeks.

barriers to recolonization by fish, than streams with lesser gradients. Families in the orders Lepidoptera, Megaloptera, and Odonata were similar but occured in such low frequencies as to render comparisons meaningless and were not shown. Several plecopterans, especially the shredders Paracapnia opis (Capniidae), Leuctra truncata (Leuctridae), Amphinemura delosa (Nemouridae), and Peltoperla cf. maria (Peltoperlidae), occur in larger percent abundances in Bruce Creek. Trichopteran numbers, overall, were similar but the net spinning hydropsichids were more abundant in Bruce Creek and the periphyton grazing Goera and Neophylax spp. (Limnephilidae) were more abundant in Crabapple Creek.

Comparing the percent composition by family of the two streams demonstrated that significant differences in the faunal composition could be overlooked by only comparing differences between orders. The limnephilid trichopterans were more plentiful in Crabapple Creek while hydropsichids were more plentiful in Bruce Creek. Collectively the Trichoptera were about equal in percent composition in the two streams. Similarly, comparisons of the percent composition by genus of the two streams also overlooks significant taxonomic differences. Collectively the heptageniid mayflies were about the same in the two streams. Referring to Table 3.1 (p. 72) reveals that the genera Cinygmula, and probably Stenonema, Stenacron, and Heptagenia occured with less relative frequency in Bruce Creek, while Epeorus occured in greater relative abundance there. All of the heptageniids were classified primarily as scrapers, except Epeorus, which was classified as a less specialized collector-gatherer (Merritt and Cummins 1984).

The ability to correlate trophic categories with relative abundances of various taxonomic groups suggested that a comparison between streams based on the functional group concept proposed by Cummins (1974) would more clearly distinguish the nature of differences in the fauna of stripmined streams. Figure 3.3 compares the percent composition of each functional group in the two streams. The functional group affiliation of each taxon was obtained from Merritt and Cummins (1978). A complete list of the functional group classification of the taxa in this study is presented in Appendix B, Table 2.

Comparing the functional group composition of the two streams reduces the differences between them to fewer ecologically important dimensions. The previously observed difference in abundances between orders and families were due to the prominence of specific functional groups within them. For example, the lower abundance of Ephemeroptera in Bruce Creek was due to the prominence of collecter-gatherers and scrapers in this order. The greater abundance of plecopterans in Bruce Creek reflected the prominence of shredders in this order. Filter feeding trichopterans (collector-filterers), primarily Hydropsychidae, were relatively more abundant in Bruce Creek. The periphyton scraping Limnephilidae were relatively more abundant in Crabapple Creek suggesting that the periphyton community of Bruce Creek was the factor limiting the numbers of these taxa there. Similarly, the periphyton grazing heptageniid genera were less abundant in Bruce Creek while the collector-gatherer Epeorus was less abundant in Crabapple Creek.

Epeorus was classified as a collector-gatherer which seems to contradict the trend for this functional group to be reduced in Bruce

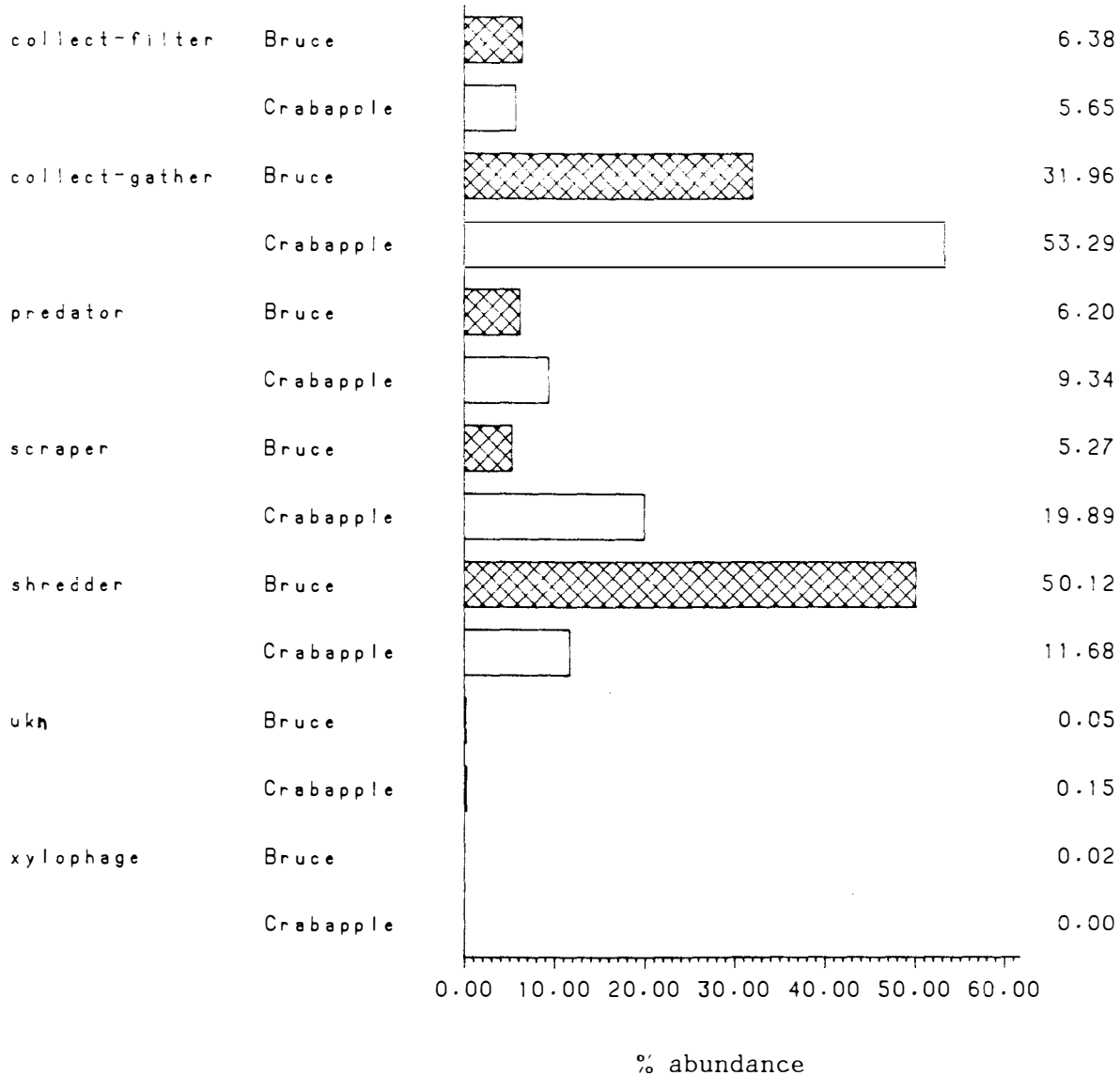


Figure 3.3. Percent of total number of individuals contributed by functional feeding groups of macroarthropods in Crabapple and Bruce Creeks.

Creek. However it is possible that Epeorus is in fact a more opportunistic feeder than most collector-gatherers and therefore prospered by the reduction of other collector-gatherers and Heptageniidae. It was also one of the few collector-gatherer mayflies which was classified as a clinger (i.e. to undersides of rocks). This may afford it some protection from surface mining effects that the other collector-gatherers lack. Minshall (1967) studied the life history of E. pleuralis in Kentucky. He described E. pleuralis as an opportunistic feeder which seemed to eat whatever plant material was available. It subsisted primarily on detritus; however, in April when the stream bottom had a rich layer of periphyton, it fed heavily on it. Epeorus dispar probably has similar feeding habits to E. pleuralis.

Matter (1978) and Matter and Ney (1980) observed similar functional group differences between mined and reference streams in southwestern Virginia. Specht et al. (1984) studied the response of functional feeding groups to effluent from the settling pond of a coal fired power plant in Adair Run, Virginia. The collector-gatherers (primarily Ephemeroptera) were reduced and the periphyton scrapers (primarily Psephenus herricki) increased. Changes in the functional group composition in this stream, however, seemed to reflect differences in physiological tolerances between different taxonomic groups. Mayflies were in general reduced, resulting in a reduction in collector-gatherers. While some mayflies are also scrappers, the increase in the coleopteran Psephenus herricki resulted in a net increase in the scraper functional group. Psephenus herricki was demonstrated to be more resistant to the heavy metals, ash, and pH conditions of the effluent than the Ephemeropteran,

Stenonema pudicum, in a bioassay experiment, further strenghtening the conclusion that Coleoptera in stripmined streams are trophically, not physiologically, limited. The Ephemeroptera seem to be more vulnerable, physiologically, to environmental degradation than any of the other major aquatic orders. However, the pollution resulting from surface mining is not characterized by toxic substances in the absence of acidification. The water quality of Bruce Creek was never as toxic as the pollution observed by Specht et al. (1984) in their study. Larrick et al. (1981) found structural and functional changes in the heterotrophic bacteria community of Adair Run in response to the same effluent events studied by Specht et al. (1981). They used assimilation of tritiated glucose to assess the functional status of heterotrophic bacteria communities. Assimilation was reduced in areas downstream from effluent emissions.

The large number of taxa and the relatively large percentage of the macroarthropod fauna of Crabapple Creek classified as scrapers emphasizes the importance of the periphyton community even in small heavily shaded streams. Fisher and Likens (1973) observed that less than 1% of the standing stock of organic matter in Bear Brook, a small stream in New Hampshire, USA was autochthonous. Since then the importance of the periphyton component of the energy base in small streams may have been underestimated. Several studies now indicate that periphyton, even the low biomass in small streams, may be very productive due partly to cropping by grazers which help to maintain the community at an immature and vigorous stage (Elwood et al. 1972, McIntire 1973, Lamberti and Resh 1983).

The lower abundances of collector-gatherers observed in Bruce Creek were probably the result of silt deposition. Silt fills the interstitial spaces of the stream bed and may inhibit the foraging of this functional group more than others. Other studies have documented that silt can cause changes in the abundance of specific taxa in streams. DeMarch (1976) found that silt was the single most important factor in determining the distribution of distinct groupings of insects in streams. Rabeni and Minshall (1977) found that a light layer of silt added to trays of coarse substrate reduced the abundance of several taxa of aquatic insects in them. Ciborowski et al. (1977) found that both silt and increased current significantly increased the drift of the mayfly Ephemerella subvaria. The breakdown of leaves was also slower on silt surfaces (Reice 1974). Unfortunately these effects have not been investigated from a functional group perspective to date.

Bruce Creek is slightly smaller than Crabapple Creek and receives a slightly greater input of CPOM (Table 2.8, p58). Smaller streams are thought to retain a larger proportion of the CPOM entering them (Bilby and Likens 1980). Bruce Creek does not have a significantly larger standing stock of CPOM meaning it must be consumed and/or transported downstream at a faster rate than in Crabapple Creek. Given its smaller size and greater inputs of CPOM Bruce Creek would be expected to have a greater proportion of shredders in its macroarthropod fauna. The absolute density of shredders was 211% greater in Bruce Creek and may partly explain the faster processing rate of dogwood leaves described in Chapter 2. Many shredders, or at least many plecopterans which dominate this functional group, are tolerant of post-mining stream conditions.

Tolbert (1978) found Amphinemura densities to be relatively unchanged in mined streams despite reductions in most other taxa. Amphinemura delosa was the numerically dominant shredder in Bruce Creek.

Conclusion

The main objective of this study was to assess the long term impacts, if any, of surface mining coal on the structure and function of macroarthropod communities in small streams in the Cumberland mountains of eastern Tennessee. Structural parameters, abundance (Table 3.1, p.70); species richness, Shannon diversity, dominance, and evenness (Table 3.2, p.88); and relative taxonomic composition of the fauna by order and family (Figure 3.1, p.81 and 3.2, p.88, respectively) were compared between the two streams. In addition, the relative functional feeding group composition of the two streams was compared (Figure 3.3, p.91) (Cummins 1974, Merritt and Cummins 1984).

Differences in the taxonomic composition of Bruce and Crabapple Creeks as a result of surface mining were similar to differences reported in some other studies. The orders Coleoptera and Ephemeroptera were the most consistently reduced in all studies. Bruce Creek also reflected this pattern. However major differences in the taxonomic composition of streams can be overlooked in comparisons at the ordinal level. In this study differences in the familial composition of Trichoptera would have been missed (Figure 3.2, p.88). At the familial level major differences in the generic composition of heptageniid mayflies would have been missed (Table 3.1, p.70). Perhaps more importantly, the differences observed at various taxonomic levels reflected

important differences in the composition of the functional feeding groups of the stream (Figure 3.3, p.88). Changes in the functional feeding group composition implied that the two streams differ in the way they utilize and channel energy and nutrients.

Bruce Creek had relatively fewer periphyton scrappers and collecter-gatherers. These patterns in functional group composition explained differences in percent abundance observed between taxonomic groups. The reduced abundance of Ephemeroptera in mined streams reflected the prominence of collector-gatherers and to a lesser extent periphyton grazers in this order. The relatively greater abundance of Plecoptera in Bruce Creek reflected the prominence of shredders in this order. Reduced abundances in Coleoptera probably reflected the reduction in Psephenus and Ectopria which are usually prominant in the beetle communities of most streams and are also obligate grazers. The higher density of shredders and faster leaf processing in Bruce Creek may also result in more fine particulate organic matter production and help explain the increased relative abundance of collecter-filterer hydropsichid trichopterans.

Vaughan (1979) demonstrated the diatom communities of mined steams in eastern Tennessee are very reduced and remain so far as long as 20 years (Vaughan et al. 1978, Vaughan 1979). The consistent reduction in Bruce Creek of periphyton grazers across taxonomic categories emphasizes the importance of the periphyton communities in small streams. This importance has, perhaps, been inadequately appreciated in the past. The results of this study indicated that the role of periphyton and autochthonous primary production in streams should be reevaluated, in

general and with regard to pollution effects. The differences in relative abundances of taxonomic and functional feeding groups may indicate that the macroarthropod communities of small streams adjust to the effects of surface mining for coal, but that some groups are much slower to recover. Comparisons of streams draining reclaimed watersheds show little biological difference from those draining unreclaimed watersheds (Matter and Ney 1981).

The results presented here also support the functional group concept of Cummins (1974) and demonstrate its value as an investigative tool in the study of lotic ecosystems. The pattern of functional group differences between Crabapple and Bruce Creeks complement the differences in taxonomic composition observed in previous studies. Taxonomic differences reflect the dominant functional group affiliation of its species and do not nessacarily indicate physiological intolerance to post-mining stream conditions.

This study does not actually support the river continuum concept of Vannote et al. (1980) and was not designed to evaluate its merits. However, the theory does help provide a perspective from which to to formulate relevant questions and future experiments. It would be most interesting to trace the effects of surface mining coal downstream through successively larger stream sizes and see if the macrobenthic communities reflect differences in the structural properties of the stream continuum which can be attributed to surface mining. There is a genuine need for such research because several of the nations most valuable river resources, the Cumberland, Tennessee, and New Rivers, to name

just a few, are increasingly influenced by the cumulative effect of surface mining in thousands of their headwater tributaries.

CHAPTER 4

SECONDARY PRODUCTIVITY OF THE MACROARTHROPOD COMMUNITIES IN A MINED VERSUS A PRISTINE STREAM IN THE CUMBERLAND MOUNTAINS OF EASTERN TENNESSEE

Introduction

The physical structure of a surface mined stream was compared to a pristine reference stream in terms of water quality and organic matter dynamics in Chapter 2. The community structure and functional feeding group composition of these streams was compared in Chapter 3. The effects of surface mining coal on the secondary productivity of macroarthropods in streams has not been previously reported. Secondary productivity of the macroarthropod community can provide a better understanding of the effects of surface mining coal than abundance data or species diversity indices because it reveals how energy and nutrient cycling pathways through the stream ecosystem are altered.

Secondary production, the biomass produced per unit area, per unit time by consumers, is an important process in any ecosystem. In stream ecosystems secondary production is important in retaining energy and nutrients within the stream. Since small woodland streams have relatively small amounts of primary production, secondary production (including heterotrophic microbes) is the main pathway by which carbon and nutrients entering the stream are retained. A stream with a great amount of secondary production should be more retentive, and have a shorter spiralling length of carbon and other nutrients than a structurally similar stream with less secondary production (Webster 1975,

1977; Newbold et al. 1981, Elwood et al. 1983). Small woodland streams such as the two studied here receive most of their carbon and other nutrients from dead and dying leaves, wood, and reproductive parts of trees. This coarse particulate organic matter (CPOM) serves as a food source for a succession of biological functional groups in the stream. Microbes attack CPOM first and render it more nutritious and palatable to macroarthropods. This process has been labeled "conditioning". Macroarthropods that chew up and "shred" CPOM are the next group to utilize it. This "shredding" process, in combination with microbial activity and mechanical fragmentation, results in the reduction of CPOM to finer particles of organic matter or FPOM (< 1 mm). This FPOM, along with algae, diatoms, bacteria, and other microscopic organism suspended in the water, is utilized by collector-filterers. The FPOM which settles to the stream bottom is utilized by collector-gatherers who forage upon the surface, under rocks, and between the interstices of the substrate for it. Periphyton grazers (scrapers) are the only functional group relatively uninvolved in this pathway. However, their activities do contribute to FPOM production via feces and dislodged periphyton. It is also important to realize that species specializing in detrital feeding may occasionally, either actively or passively, consume plant or animal material. Some species are known to become increasingly predaceous as they grow. I have noticed preserved specimens of Yugus / Malirekus spp. with detritus and Chironomids in their mouths on separate occasions. The taxonomic and functional group differences between Crabapple and Bruce Creeks reported in Chapter 2 suggest that specific energy and nutrient pathways in a stream are disrupted by the effects of

surface mining coal. Estimating the relative production of specific taxa and functional groups in the two streams is the most direct means of assessing these effects.

The secondary productivity estimates reported here are not absolute annual estimates. They are only valid for the period of sampling, January through May, 1984. They are not corrected for cohort production interval which is used to adjust annual production estimates to the amount of time it actually takes the species to develop from hatching through pupation or emergence. The period of sampling in this study did include a significant proportion of the cohort production interval of some species. In particular two of the dominant aquatic insects, Ephemerella dorothea and Amphinemura delosa were collected over most of their active growth period in the streams. However the reader should realize that the production values reported here would be considerably different if an entire year where sampled. Most values would probably increase so these values can be considered conservative estimates. The values are most useful as comparative indices of energy use by different functional compartments in two streams.

The effects of clearcut logging on the secondary productivity of Peltoperla maria was reported by O'Hop et al. (1984). The secondary productivity of a species depends on the initial abundance of the cohort, the survivorship over time, and the average size attained at maturity and voltinism (Waters 1979). To an extent production of species in other streams can be inferred from the production estimates for the same species in Crabapple and Bruce Creeks by comparing their abundances. However O'Hop et al. (1984) found similar production of

Peltoperla maria in two streams with different abundances of this species. Larger size and greater survivorship offset the difference in abundance between the two streams.

Detailed reviews of the various methods of estimating secondary productivity have been written by Waters (1977) and relative to aquatic insects in particular by Benke (1984). Four main approaches to estimating productivity have been developed. Removal-summation methods sum the losses of biomass of individuals in the cohort between successive samples, taking into account the sizes at loss. Waters (1977) attributes the development of this method to Boysen-Jensen (1919), Anderson and Hooper (1956), Sanders (1956) and Teal (1957). An alternate method, the instantaneous growth method, sums the product of instantaneous rate of growth and the mean standing stock of that time period, over the several sampling periods covering the species growth (Ricker 1946, Allen 1949). Allen (1951) extended the instantaneous growth method to a graphical representation, the Allen curve, where density in numbers is plotted against mean weight. The area under the curve, expressed in appropriate units is the cohort production. The removal-summation and Allen curve methods are most applicable to organisms which can be identified by age and cohort, such as fish. Hynes (1961) proposed an alternate approach and later a refinement (Hynes and Coleman 1968), which essentially substituted size classes for age classes in the removal summation method. Hamilton (1969) corrected additional errors and the method became established. Originally known as the Hynes method and later the Hynes-Hamilton method, it is now generally referred to as the size-frequency method.

Waters and Crawford (1973) compared the four methods using data on Ephemerella subvaria and found the size-frequency method yielded higher estimates. Cushman et al. (1977) compared the four methods with a computer simulation, evaluating the effect of growth curve and sampling regime on them. They found estimates qualitatively similar to those of Waters and Crawford (1973) but concluded that the Hynes method was at least as accurate as the instantaneous growth method. O'Hop et al. (1984) using data for Peltoperla maria in two different streams, Resh (1977) using data for Ceraclea ancyllus, and Cushman et al. using data for Diplectrona modesta and Alloperla mediana found the different methods to yield comparable estimates of secondary productivity. The validity of the size-frequency method is now generally accepted and numerous studies have utilized it in the estimation of secondary productivity (Mclure and Stewart 1976, Martien and Benke 1977, Benke and Wallace 1980, Bright 1982, Garmon and Waters 1983, Benke et al. 1984, Freeman and Wallace 1984, Jop and Szczytko 1984, Ladle et al. 1984, Jackson et al. 1985).

Krueger and Martin (1980) developed a way to compute confidence intervals for the size-frequency estimates allowing statistical inference to be made concerning their accuracy and differences. The estimation of confidence intervals has subsequently been extended to the other methods of estimating secondary productivity (Newman and Martin 1983).

Materials and Methods

Physical and chemical water properties were measured as described in Chapter 2. The sampling, processing and identification of aquatic macroinvertebrates was described in Chapter 3. The total length (exclusive of cerci) and width across the eyes of all the macroinvertebrates was measured to the nearest .025 mm using a Nikon dissecting scope with an ocular micrometer. The secondary productivity of each species which represented at least 1% of the fauna during the sampling period was calculated by the size-frequency method (Hynes and Coleman 1968, Hamilton 1969) as modified by Krueger and Martin (1980) for obtaining variance of the estimate which allows for statistical inference. Species under 10 mm were assigned to 1 mm size classes, larger species were assigned to 2.5 or 5 mm size classes.

Specimens for the length-weight regressions were obtained from several local streams (Knoxville, TN) by qualitative sampling. The specimens were returned to the laboratory and measured alive. Then about half were placed in a drying oven at 60 ^{o} C for 24 hours and dessicated in a vacuum jar with drierite for an additional 24 hours. The other half were preserved in 80% ethanol for at least 30 days before drying and dessicating the same as the first group. This preservation group allowed the length-weight regressions obtained from the fresh killed specimens to be corrected for preservation when applied to the samples from Crabapple and Bruce Creeks. Individuals from both groups were weighed to the nearest ± 1 µg on a Kahn model 7000 electrobalance.

A SAS computer program (SAS Institute 1982) was used to calculate secondary productivity and estimate length-weight regression equations

for each taxa. Length-weight regression relationships which were not estimated directly were obtained from Smock (1980) who obtained very similar results to mine for taxa we both analyzed.

Results and Discussion

The size-frequency method of estimating secondary productivity assumes that length corresponds roughly to age and that a reliable relationship between length and weight can be determined for the species concerned. Considerable variation exists in the methodology used to derive length-weight equations between studies. This has inhibited comparisons of secondary productivity for the same taxa between studies (Resh 1977). Some studies have used length versus wet weight and others have derived regressions of length versus ashed weights. I believe the use of dry weights as reported here and by Smock (1980) provides the most precise length-weight estimates for the amount of effort. The weighing of individual animals also provides more information on the inherent variation of weight with length. Use of wet weight is subject to error in the amount of moisture retained by the specimens, and centrifugation to minimize excess water might also cause loss of haemolymph. Ashing individuals reduces their weight to such an extent that measurement errors are an order of magnitude greater.

It has been empirically determined that a power function of the form:

$$Y = \alpha X^{\beta}$$

provides the best description of weight (Y) as a function of length (X)

for both terrestrial (Rogers et al. 1976) and aquatic (Smock 1980) insects. This function can be linearized by log transformation to:

$$Ln(Y) = Ln(\alpha) + \beta * Ln(X)$$

allowing linear least squares regression to be used to estimate the intercept, $ln(\alpha)$, and slope, β, parameters of the regression line.

Table 4.1 contains the regression coefficients and sample size of each length-weight equation to be used in the estimation of secondary productivity for the species (taxa) which contributed at least 1% to the numerical abundance in either stream. The length-weight equations for a few taxa were obtained from the literature because I didn't collect enough individuals in these or closely related taxa for a reliable regression analysis. All equations obtained from the literature were obtained from Smock (1980), except the equation for elmid beetles which was taken from Benke (1984), because he used methods most similar to mine. The methods used by Smock (1980) were essentially the same as I used only he froze his specimens prior to measuring and weighing. Statistically similar equations were obtained for several taxa which we both analyzed. The similarity between our results demonstrates the accuracy and reproducibility of our approach (i.e. dry weights of individuals regressed against length). Table C1, Appendix C contains an extended table of length weight regressions, including the 95% confidence intervals for the estimates, coefficient of determination (r^2), and sample size for additional taxa which were not included in the productivity estimates. This table includes values obtained in this study and from the literature, primarily Smock (1980).

Table 4.1. Regression coefficients of length-weight equations for the taxa which contributed at least 1% of the abundance in either Crabapple or Bruce Creek.

Taxa	Ln (α)	ß	N
Coleoptera			
Elmidae			
spp.[1]	-5.66	3.01	-
Eubriidae			
Ectopria nervosa	-5.26	3.48	23
Psephenidae			
Psephenus herricki	-6.16	3.79	11
Ephemeroptera			
Baetidae			
Baetis brunneicolor[2]	-5.52	3.02	20
Ephemerellidae			
Ephemerella dorothea	-5.12	3.03	66
Heptageniidae			
Cinygmula subaequalis	-4.62	2.71	29
Epeorus dispar			
Leptophlebiidae			
Habrophlebiodes americana	-5.80	3.09	11
Siphlonuridae			
Ameletus lineatus[2]	-5.52	3.02	20
Isopoda (Crustacea)			
Asselidae			
Lirceus fontinalis	-4.76	2.84	24
Plecoptera			
Chloroperlidae			
Hastaperla brevis[3]	-5.28	2.84	16
Leuctridae			
Leuctra truncata[3]	-5.28	2.84	16
Nemouridae			
Amphinemura spp.	-4.97	2.68	10
Peltoperlidae			
Peltoperla cf. maria	-4.88	2.08	17
Trichoptera			
Hydropsichidae			
Diplectrona modesta	-5.81	2.78	15
Limnephilidae			
Neophylax spp.[4]	-5.04	2.74	16

[1] From Benke et al. (1984)
[2] regression derived for *Isonychia spp.*
[3] from Smock (1980), regression derived for *Allocapnia spp.*
[4] from Smock (1980), regression derived for *Pycnopsyche guttifer*

The regression coefficients in Table 4.1 include pooled estimates for some families along with the coefficients for particular genera if available. For example, the regression coefficient for Leptophlebiidae were derived from 11 individuals of Habrophlebiodes and 17 Paraleptophlebia. The data for the two species were pooled because a much wider size range was thereby included in the regression. This helps to prevent bias due to underrepresentation of large or small size classes. The regression coefficients for Epeorus dispar and Cinygmula subaequalis were derived from 29 individuals of at least two species: E. rubidus/subpallidus and E. pleuralis. The productivity estimates for Cinygmula subaequalis are based on the regression for Epeorus because not enough Cinygmula were found to perform a separate regression for this genus. Individuals of Cinygmula subaequalis seemed to resemble Epeoris dispar morphologically more than most of the other Heptageniidae. In cases where a regression was obtained from the literature it was often only an approximate estimate for the taxa concerned. For example, productivity estimates for both the Leuctra and Hastaperla were based on regression coefficients from Smock (1980) for Allocapnia spp. which is another long and slender plecopteran. Likewise the estimates for the Neophylax are based on Smock's equation for Pycnopsyche guttifer the only other limenphilid for which a length-weight relation was derived. The slope coefficient, β, for Peltoperla cf. maria seems a bit shallow but the specimens spanned a reasonable size range and no other source of error is apparent. The unusual morphology of this genus must be responsible.

Figure 4.1 is a regression plot of the natural logarithm of weight as a function of the natural logarithm of length based on data from

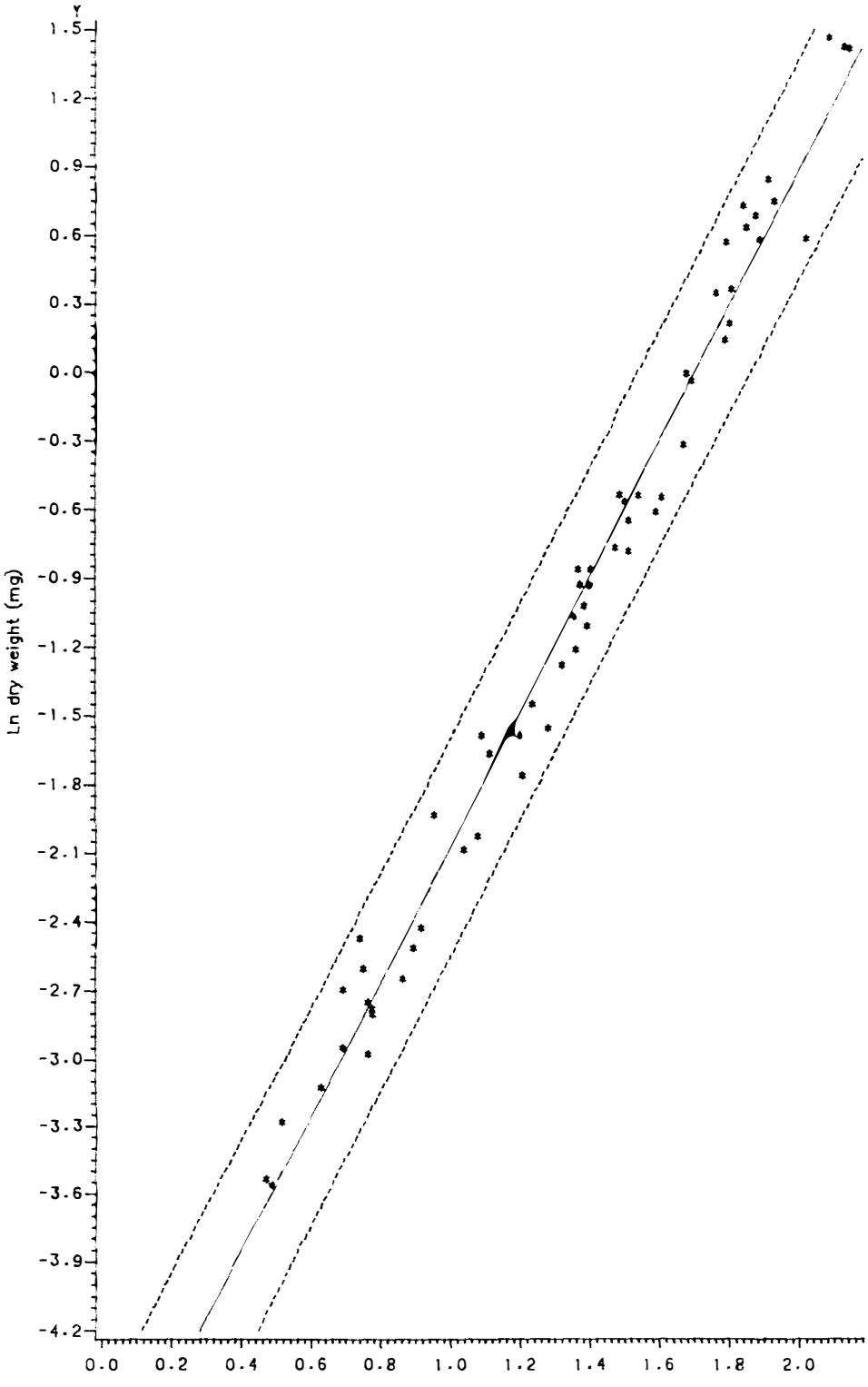


Figure 4.1. Plot of weight as a function of length based on a regression equation for several species of Ephemerella mayflies.

several species of Ephemerella mayflies. The species known to be included in the regression are E. dorothea, E. invaria, E. hispida and E. rotunda but other species may also be present. As can be seen from the plot most species of Ephemerella mayflies are isomorphic, i.e. they are described by the same allometric equation. Morphologically distinct species such as Ephemerella septentrionalis may not fit this curve. Eurylophella spp. were not significantly different from Ephemerella spp. based on the results from Smock (1980) (Appendix C, Table C1). Drunella spp. were not estimated by myself or Smock in sufficient numbers to obtain a reliable regression, but a priori, I would expect them to be more similar to Heptageniids than Ephemerellids, based on their morphology. Most species within a genus, and probably most genera in a family can be described rather accurately by the same length-weight equation.

While the length-weight summary presented here is one of the most complete descriptions available to date for aquatic insects, there is still a need for additional work on these and other species. Resh (1977) has pointed out that it is very difficult to compare results from various productivity studies because of differences in the derivation of length-weight equations. Ideally, a set of standard equations, along with information on the size range, sample size, and confidence intervals for each, can be compiled for a wide range of species so that this source of variation is eliminated.

As seen from Figure 4.1, the logarithmic transformation of length and weight still does not produce perfect linearity in the data, at least for most species. This result makes it especially important that

length-weight regressions be based on a complete size range of individuals for each taxon. This condition was not fully met for all of the taxa reported here, and I hope additional observations will eventually be added to these data. If a regression does not include individuals at one end of the size range of the taxa it will be pulled up or down by the other end. For example, if small Ephemerella had not been obtained for the regression in Figure 4.1 the slope of the line would have been steeper. Since these are logarithmic coordinates the error would be magnified in the original units. For example, if the slope were estimated to be only 10% greater, say 3.3 instead of 3.0, then the difference in weight estimates for a 9 mm specimen would be 8.42 versus 4.36, respectively. A difference of almost 100%. The error in productivity estimates can be even greater since the larger size classes may contribute a large fraction of the total productivity.

It is not possible to summarize the calculation of production estimates and their variances in a single equation. For a thorough algebraic description of the calculations used to derive production estimates in this dissertation see Krueger and Martin (1980). Table 4.2 demonstrates calculation of production for Ephemerella dorothea by the Size-frequency method as modified by Krueger and Martin (1980). In this calculation: N is the estimate of the number of individuals (m^{-2}) on the average during the sampling year for each size class, v(N) is the variance of this average, B is the average biomass (mg/m^{-2}) (N x average weight of individuals in this size class), Number lost is the difference between N in size class i and size class i+1, Mean weight is the mean weight per individual in size class j, Weight loss is the geometric mean

Table 4.2. Calculation of Ephemerella dorothea production (mg/m^2 dry) by the Size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	X Number size classes (mg/m^2)
				Crabapple Creek			
0-2	175.58	1473.21	5.27				
				53.95	0.05	2.64	21.14
2-3	121.63	716.19	9.73				
				88.09	0.14	12.70	101.64
3-4	33.54	56.55	8.72				
				5.13	0.37	1.90	15.23
4-5	28.41	43.41	15.06				
				18.84	0.72	13.58	108.62
5-6	9.57	4.11	9.38				
				2.48	1.27	3.14	25.15
6-7	7.09	6.95	11.63				
				4.72	1.98	9.34	74.76
7-8	2.37	0.22	5.66				
				1.86	2.84	5.28	42.23
8-9	0.51	0.11	1.72				
				0.51	1.84	0.94	7.49
	total biomass=67.16				total production=396.27		
					variance=(1489.47)		
				Bruce Creek			
0-2	40.02	23.90	1.20				
				-11.52	0.05	-0.60	-4.79
2-3	51.54	30.58	4.64				
				33.15	0.15	4.97	39.78
3-4	18.39	11.34	4.60				
				7.77	0.38	2.93	23.46
4-5	10.62	8.15	6.05				
				8.32	0.73	6.06	48.46
5-6	2.30	0.62	2.14				
				1.42	1.19	1.69	13.55
6-7	0.88	0.20	1.35				
				0.03	2.02	0.06	0.49
7-8	0.85	0.15	2.27				
				0.60	3.04	1.82	14.59
8-9	0.25	0.06	0.86				
				0.25	1.86	0.47	3.72
	total biomass=23.11				total production=139.26		
					variance=(157.26)		

$(W_j W_{j+1})^{.5}$ of the mean weights per individual of size classes j and j+1, the final column is production of each size class which is simply weight loss multiplied by the number of size classes. The sum of this column is the total production for the species over the time period. The v(N) column is also summed to yield the variance of the total production. This variance component can be used to derive an approximate 95% confidence interval which represents 2 SE on either side of the production estimates as follows:

$$\text{Production} = \quad \pm 2 * [(\text{variance})^{0.5}]$$

Table 4.3 presents the standing stock (B) and production estimates (P) for each of the taxa which contributed at least 1% of the numerical abundance in Crabapple or Bruce Creeks. The calculation tables for these production estimates for each of these taxa as demonstrated for Emphemerella dorothea, Table 4.2, are in Appendix C, Tables C2-C20.

As can be seen from Table 4.3 the productivity of many of these taxa varies by a factor of two or more between streams. All of the productivity estimates were significantly different from zero. Only Baetis brunneicolor, Ameletus lineatus, Peltoperla cf. maria, and Neophylax cf. mitchelli were not significantly different, i.e., did not overlap each other using approximately 95% confidence intervals. Although the productivity of these taxa was not significantly different they differed in a manner consistent with the general trends. For example, all the ephemenopteran's except Epeorus dispar had lower productivity in Bruce Creek while Peltoperla cf. maria had higher productivity in Bruce Creek, as did the other plecopteran shredders, except A. wui. The situation for A. wui is interesting because it is more numerous and

Table 4.3 Summary of standing stock (B), production (P) and variance of the production estimate for each taxon which contributed at least 1% of of the numerical abundance to either Crabapple or Bruce Creeks.

Taxa	Crabapple Creek			Bruce Creek		
	B	P	Var	B	P	Var
Elmidae spp.	4.7	20.8	6.5	1.1	3.3	4.4
Ectopria nervosa	30.4	141.0	294.4	12.0	28.7	45.6
Psephenus herricki	31.6	147.9	1388.3	8.5	30.2	116.4
Simuliidae spp.	9.6	39.0	32.3	5.7	18.9	19.0
Hexatoma spp.	62.2	358.4	1603.8	19.8	98.8	537.4
Baetis brunneicolor	9.4	37.9	27.0	7.0	32.4	16.0
Ephemerella dorothea	67.2	396.3	1489.5	23.1	139.3	157.3
Cinygmula subaequalis	14.1	64.3	81.5	.7	2.5	.9
Epeorus dispar	45.2	288.4	725.0	93.1	420.8	2377.1
Habrophlebiodes americana	16.1	71.6	86.8	1.3	4.8	3.3
Ameletus lineatus	92.6	413.8	1209.2	87.8	371.6	2328.8
Lirceus fontinalis	34.5	136.4	1106.3	246.3	1388.1	8442.4
Hastaperla brevis	9.6	44.1	38.1	3.2	13.7	5.9
Leuctra truncata	1.7	7.0	2.4	3.3	19.7	26.7
Amphinemura delosa	2.5	11.4	1.8	13.0	62.8	31.1
A. wui	5.7	25.0	14.8	2.8	12.6	3.0
Peltoperla cf. maria	8.5	38.9	95.6	12.2	65.8	185.1
Diplectrona modesta	9.9	40.7	39.2	30.8	107.5	218.4
Neophylax cf. mitchelli	10.6	36.5	104.8	5.9	21.1	42.9
N. spp.	30.9	143.3	331.5	11.6	45.4	91.5

productive in Crabapple Creek, yet it is also a common inhabitant of rip-rap habitat in reservoirs in lower elevations in this region (Etnier, pers. comm.). Another interesting feature in this data is the reversal in productivity between the two streams for the closely related congeners A. wui and A. delosa. Neophylax cf. mitchelli is the only other species without significantly different productivity between the two streams. The confidence interval for this species was especially large, probably resulting from its highly aggregated spatial distribution because it is found almost exclusively on large rocks. More samples at more frequent intervals may have reduced the confidence interval for Neophylax cf. mitchelli productivity estimates sufficiently to resolve a significant difference between streams since this species seemed to be twice as abundant in Crabapple Creek.

Table 4.4 compares the productivity (P) and standing stock (B) of functional groups in Crabapple and Bruce Creeks. The xylophage functional group is not included because it contained only one individual. Except for Hexatoma spp, most of the predators were scarce and/or semivoltine so that only a small part of their life cycle could be sampled. This includes the Megaloptera, Odanata, and all the perlid and perlodid stoneflies. If more sample dates were available it would probably be possible to obtain production estimates for the most abundant perlodids, Yugus / Malirekus, and the perlid Acroneuria carolinensis. They were, however, similar in numerical abundance in the two streams. Williams (1981) observed an interesting situation in the abundance of perlid and perlodid stoneflies in the four streams she studied. One on the mined streams, Dry Branch, contained no perlids. The other three streams in

her study contained both taxa and two of them (both first and second order streams) contained very similar abundances of the two taxa, which is the situation in Crabapple and Bruce Creeks. However, the number of perlodids in Dry Branch was comparable to the combined abundances of perlids and perlodids in the other streams. There was not a large difference in the overall abundance of predators between the two streams studied here except for the Hexatoma spp.

Table 4.4. Productivity (P) and standing stock (B) of functional groups in Crabapple and Bruce Creeks.

	Crabapple		Bruce	
Functional group	P	B	P	B
collector-filterer	79.73	19.51	126.45	36.48
collector-gatherer	1255.23	240.16	982.53	215.57
predator	358.37	62.24	98.82	19.76
scraper	553.89	122.32	133.89	39.80
shredder	227.57	52.78	1544.07	277.55
total	2474.79	497.01	2885.76	589.16

The overall productivity of Bruce Creek is actually greater than for Crabapple Creek. The difference is not as great as the totals of Table 4.4 suggest because a larger proportion of the total fauna of Bruce Creek than of Crabapple Creek is included in the total. This is because Crabapple has many more species which occur at lower frequencies than does Bruce Creek. Species occurring at low frequency are not included in the analysis because of the difficulty in obtaining reliable production estimates on small cohort sizes. In addition a large portion

of the total productivity in Bruce Creek is contributed by Lirceus fontinalis, a crustacean. If the difference in productivity between Crabapple and Bruce Creek due to L. fontinalis is removed, the overall productivity of the insect community in Bruce Creek is only 60% of Crabapple Creek. The question is if the abundance of L. fontinalis in Bruce Creek is natural for this stream or if it represents a form of compensatory release.

It was shown in Chapter 1 that Bruce Creek receives a greater input of CPOM (Table 2.3, p.44) and should, therefore, be able to support a larger macroarthropod community. However if the insect community of Bruce Creek were to be as productive as in Crabapple Creek, including the productivity of L. fontinalis, the productivity of Crabapple Creek would be only 64% of the productivity of Bruce Creek. It seems unlikely that the productivity of Bruce Creek should be that much greater than Crabapple Creek were it also a pristine watershed. I believe the disproportionate abundance of L. fontinalis in Bruce Creek is probably a response to the reduction of other groups in the stream. If this is true it implies that lotic ecosystems are homeostatic with regard to production and standing stock but unstable in taxonomic composition. This is not an altogether surprising result, given the high degree of functional and ecological redundancy in the fauna of streams.

Given the relatively high productivity of Bruce Creek does this mean that it has adjusted to the altered hydrology due to mining and how permanent is this taxonomic and functional group composition? If energy is flowing through the consumer compartments of Bruce Creek as well as that in Crabapple Creek, does this then inhibit the taxonomic recovery

of Bruce Creek? The evidence suggests that, as the time since mining increases, the taxonomic composition of streams becomes increasingly similar to unmined streams (Talak 1977, Vaughan et al. 1978). Figure 4.2 compares the % of productivity by functional groups between Crabapple and Bruce Creeks. Their are large differences in % productivity of functional groups between Crabapple and Bruce Creeks in all but the collector-filterer functional group. The influence of L. fontinalis increases the contrast between the two streams. If Crabapple and Bruce Creeks contained proportional numbers of L. fontinalis then the differences in the other functional groups, except collector-filterers, would be reduced. Collector-filterers would represent twice the % of productivity in Bruce Creek compared to Crabapple Creek but collector gatherers and scrapers would be almost comparable. However the fauna of Bruce Creek would undoubtedly be different in other ways if it did have similar numbers of L. fontinalis.

Conclusion

The effects of surface mining coal on the community structure of small streams in the Cumberland mountains of eastern Tennessee and in other areas of the Appalachian coal province (in the absence of acid mine drainage) have been characterized by Talak (1977), Tolbert (1978), Williams (1981), Vaughan et al. (1978, 1979, 1982) and the present study. All have observed reductions in the abundance of aquatic macroarthropods in mined streams. The reduction in abundance is most acute in the first few years following mining and gradually diminishes.

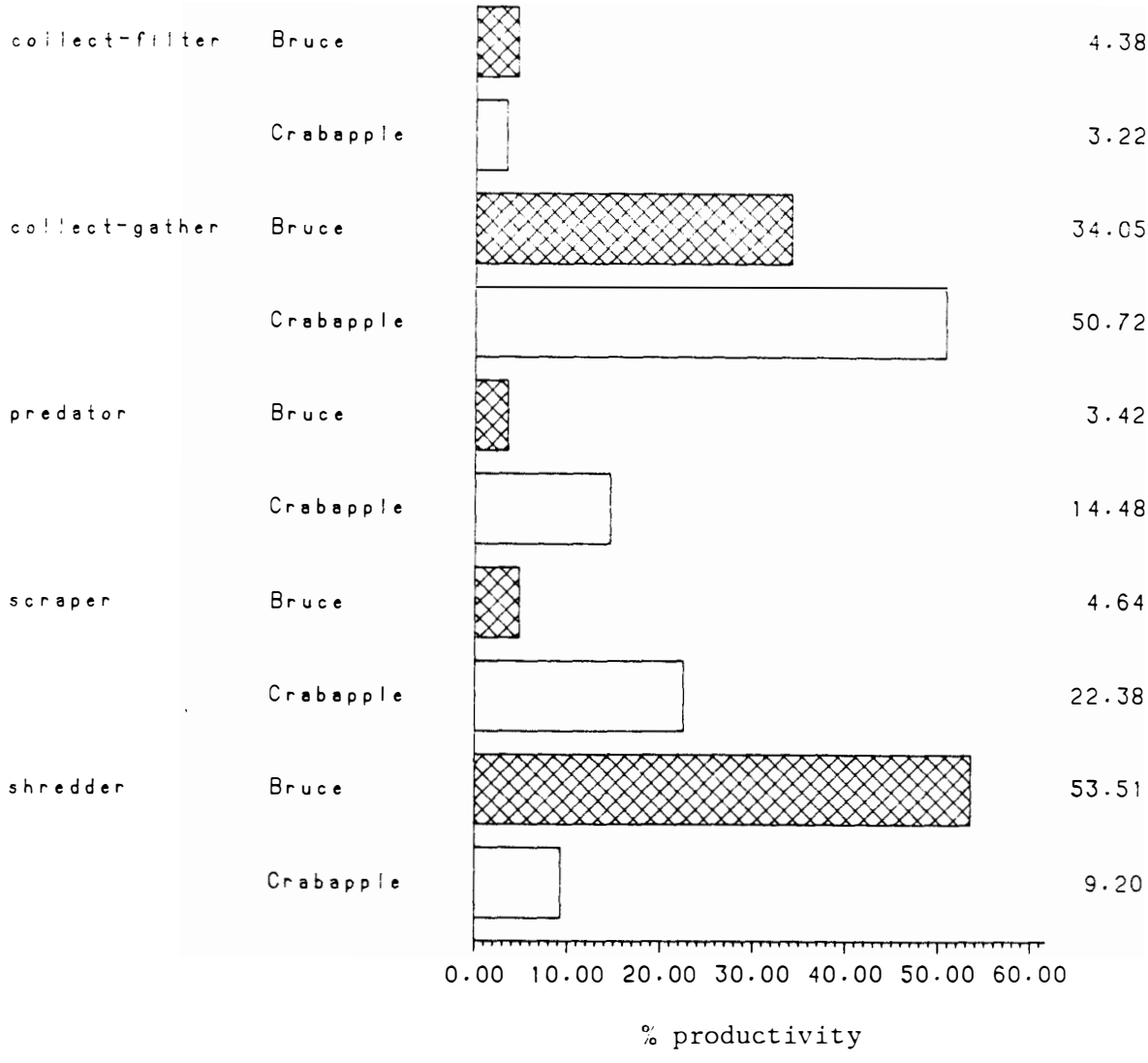


Figure 4.2. Percent of total secondary productivity of Crabapple and Bruce Creeks by functional groups.

Talak (1977), Tolbert (1978) and Vaughan et al. (1978) also noted a reduction in Shannon diversity which followed the same recovery pattern as abundance. Williams noted that diversity in two streams mined over twenty years prior to sampling was comparable to the two reference streams, confirming that diversity recovers. She also found significantly lower abundances in the macroarthropod communities of the two mined streams leading her to speculate that productivity of these streams was still impaired after more than twenty years. The mined stream in this study had generally lower diversity values than the control stream, but only slightly. In fact the diversity values from Bruce Creek might compare favorably with some pristine streams, also confirming that diversity recovers. The abundance of macroarthropods in Bruce Creek was also less than in Crabapple Creek, supporting William's findings. However the actual total production is probably greater in Bruce Creek than in Crabapple Creek due largely to the disproportionate contribution of L. fontinalis to productivity in this stream. Thus, productivity may not be proportional to abundance. This result demonstrates the principle that it is not always possible to extrapolate from one organizational level to another, i. e. population to community.

The taxonomic differences observed by Talak (1977) and Vaughan et al. (1978) in streams mined over twenty year prior to sampling relative to reference streams mean that productivity comparisons between these streams cannot be inferred from the abundance of their macroarthropod faunas. One of the important findings of this study is that productivity should be estimated directly when the taxonomic composition of streams differs. Even when the abundance of a species differs between

streams, their productivity may not. This is the case for Peltoperla maria in Coweeta, North Carolina streams (O'Hop et al. 1984). This situation occurs because productivity depends on survivorship, growth rate, and ultimate size as well as density. The use of indices of community structure, such as Shannon diversity, has occupied a prominent role in the analysis and description of the ecological effects of surface mining coal and other sources of disturbance to lotic ecosystems. However, in the final analysis one must conclude that diversity indices, though reliable monitoring tools, only partially describe the causal relationships between the disturbance and its biological results. Cummins (1974) was perhaps the first to call on the community of aquatic ecologists to explore a more functional approach to the study of problems in lotic ecology. His advice has proven valuable in this case. The two streams studied here showed clear contrasts in their functional organization. These two streams in themselves are not an adequate sample upon which to generalize, but the observations of the taxonomic composition and abundances in many other mined streams are consistent with the kinds of functional group differences observed here.

The functional group approach applied here has helped to refine our understanding of the effects of surface mining on aquatic macroarthropod communities. The effects of surface mining coal, based on this study, can be summarized as follows. The energy pathways through the various consumer groups of the stream are altered, and the total energy flow through all consumers is reduced but can recover before diversity and abundance. The recovery of productivity before diversity, abundance, and taxonomic composition probably reflects a large amount of functional

redundancy in the fauna of stream ecosystems. The insects, and presumably much of the other fauna, that depends on the periphyton community (scrapers) are among the most reduced in mined streams. Vaughan (1979) observed that the diatoms, an important component of the periphyton community are drastically reduced in mined streams, and one of the last communities to recover. These results emphasize the importance of and need for additional research into the effect of surface mining on the microbial and periphyton communities of lotic ecosystems. Collector-gatherers, usually the dominant functional group in undisturbed forest streams, and comprised largely of mayflies, are also reduced in mined streams. This result is probably the direct consequence of silt deposition. This silt may fill much of the interstitial spaces in the stream bed where these organisms forage, or otherwise inhibit their feeding. Many insects have been observed to increase their rate of drifting in response to siltation, but this is only a proximal explanation of their response. There must be some ultimate reason to cause them to respond this way and this study suggests that feeding inhibition especially of collector-gatherers is a likely possibility. It is also important to keep in mind that this is a minimally disturbed stream compared to most streams affected by surface mining and that this is a relatively long-term effect under study (circa 15 years). The results from this study may not apply to more intensively mined or even typically mined streams by contemporary standards.

CHAPTER 5

SYNTHESIS

Overview

The main objective of this study was to relate the long term impact of surface mining coal on the physical structure of small streams in the Cumberland mountains of eastern Tennessee to the structure and function of their macroarthropod communities. Bruce Creek, with approximately 3% of its surface area disturbed by mining 15 years prior to the study, and Crabapple Creek, a pristine reference stream, were compared. The % of watershed area disturbed in Bruce Creek is small compared to most surface mined streams. In addition, most surface mined streams have recovered a significant amount of the biological diversity and abundance found in unmined streams within fifteen years after mining. Nevertheless, Bruce Creek displays several physical and biological features typical of mined streams, indicating that it is still significantly affected by the mine disturbance it experienced. Bruce creek has water quality parameters characteristic of mined streams. The higher SO_4 concentrations, hardness, turbidity, and suspended solids of the water in Bruce creek indicate the watershed is still influenced by mine spoil. Another factor is the lower species richness and diversity of Bruce Creek. The species richness may be more diagnostic than the diversity in this case since the diversity of Bruce Creek is actually fairly high. Rare species have a modest effect on diversity indices, but tend to be slow to return to disturbed streams (also noted by Williams (1981)). The taxonomic composition of Bruce Creek is similar to other mined streams

studied in this area, particularly with respect to the relatively small percentage of Coleoptera and Ephemeroptera in its fauna.

Siltation and altered hydrology are two effects of surface mining on the physical structure of streams which have been well documented. Silt fills pools and the interstices of the stream bed resulting in increased peak flows due to decreased stream capacity (and increased runoff from spoil banks). Silt might interfere with the trophic ecology of some macroarthropods by reducing periphyton and burying detritus. The effects of siltation and increased peak flows on the carbon balance of surface mined streams has not been previously documented. It would seem that silt could effect processing rates of CPOM, and increase the transport of POM through the stream. Little evidence was found to support the theory that silt and increased flows altered the processing rate or transport of POM through the stream. Bruce Creek, the siltier of the two streams, had faster processing rates for dogwood leaves than Crabapple Creek. This faster processing rate is at least partially attributable to a greater density of macroarthropod shredders, but differences in the physical structure of the stream may have also contributed. For example, if much of the CPOM sampled in Bruce Creek is unavailable to macroarthropod shredders because it is trapped in silt, then the leaf bags used to estimate processing rate would have experienced a greater relative density of shredders than leaf bags in Crabapple Creek. Studies on streams with more typical amounts of watershed surface area disturbance may yet reveal surface mining effects on the POM dynamics of streams.

Streams draining surface mined watersheds undergo dramatic changes in the structure of their biological communities. Abundance, species richness, and diversity of diatoms, fishes, and invertebrates decline immediately following the onset of mining. Except for diatoms, they begin to recover within a few years after its termination in the absence of acid mine drainage. In at least some streams the diversity and species richness of mined streams recovers faster than abundance (Williams 1981). This was close to being the case in Bruce Creek as well. The Shannon diversity of Bruce creek macroarthropods was lower than, but close to that for Crabapple Creek. The density and species richness of Bruce Creek macroarthropods was, however, clearly less than Crabapple Creeks.

Indirect evidence supporting the theory that siltation interferes with the trophic ecology of certain macroarthropod functional groups was obtained through this study. Specifically, those organisms which feed on periphyton (grazers) or fine detritus particles on or in the stream bed (collector-gatherers) are the most reduced in abundance in mined versus reference streams. Most predators occured in similar densities in the two streams, but collector-filterers and especially shredders were more abundant in Bruce Creek.

Intuitively it would seem that streams with less macroarthropod abundance would be less productive. However Bruce Creek had similar, if not greater, productivity than Crabapple Creek with fewer macroinvertebrates. This is because larger species such as Lirceus fontinalis can accumulate an amount of secondary productivity comparable to a much larger number of smaller species. This study indicates productivity can

recover before abundance, and probably before diversity. Results from Bruce Creek suggest recovery of production may occur in 10-15 years whereas recovery of original taxonomic composition may require considerably longer than twenty years. The productivity of the various functional groups in the two streams reflected their functional group composition. That is productivity of grazers and collector-gatherers was lower in Bruce compared to Crabapple Creek, while the opposite was true for collector-filterers and shredders.

Comprehensive Summary

Chapter 2 compared some aspects of the organic matter dynamics between the two streams. Differences in the organic matter dynamics of the two streams are assumed to reflect differences in the physical and/or biological structure between the streams. Input, standing stock, some limited aspects of transport, and relative processing rates of two leaf species were compared. Inputs and standing stock of POM were not significantly different between the two streams. Crabapple Creek had significantly more FPOM but contamination of the samples by fine coal particles may have affected its measurement. There was no significant difference in the transport of POM as measured by drift net but this was a very imprecise method of estimating transport and sampled only a narrow range of flow conditions. There was a significant difference in transport of very fine suspended matter as measured by suspended solids analysis of water samples. This suspended solids analysis was also contaminated by coal and so should be interpreted cautiously. The processing rate of dogwood leaves was faster in Bruce Creek, but no differ-

ence was observed in the processing rate of beech leaves. Dogwood leaves are more nutritious and palatable than beech leaves. They are probably consumed by macroarthropod shredders with relatively little microbial conditioning. The beech leaves probably had not been conditioned by bacteria and fungi long enough to make them attractive to macroarthropods. The discrepancy between input and standing stock of POM and the faster processing rate of dogwood leaves in Bruce Creek indicated that it was functionally different from Crabapple Creek. The faster processing rate of dogwood leaves in Bruce Creek might be the result of higher absolute shredder densities and greater processing capacity in Bruce Creek. However, while the organic matter dynamics of the two streams are certainly different, and the functional group composition of the two streams is partly responsible, physical differences between the two streams probably also contributes. These results point out the need for complete carbon budget studies on mined streams.

A detailed description of the macroarthropod communities of the two streams (Chapter 3) revealed structural and functional differences. Structurally the fauna of Crabapple Creek was richer, more diverse, and more abundant than the fauna of Bruce Creek. The difference in diversity was small but consistent. The richness (number of species) was more disparate, reflecting the tendency of mined streams to have fewer rare species. The two streams differed functionally in that collector-gatherer and scraper functional groups were relatively more abundant and productive in Crabapple Creek and shredders were relatively more abundant in Bruce creek. Hexatoma spp., a tipulid predator, was more abundant in Crabapple Creek, but other predators were evenly represented

between both streams. The Chironomidae and Ceratopogonidae were not analyzed quantitatively. The taxonomic richness (32 vs. 20) ascertained from qualitative sampling, was greater for Crabapple Creek and suggests that the abundance may have also been greater. If so, this may partly account for the difference in the Hexatoma spp. abundance. This study reinforces the conclusion that functional as well as structural properties of stream communities are affected by surface mining. Differences in the functional group composition of the two streams provide additional insight into the nature of surface mining effects on aquatic macroinvertebrates. These differences suggest that trophic factors, reduction of primary productivity, and inhibition of collector-gatherer feeding contribute to or cause the taxonomic changes observed in surface mined streams.

Chapter 4 compared the the overall productivity, the productivity of the twenty numerically dominant taxa, and the productivity of the major functional groups in the two streams. Bruce Creek had greater overall productivity than Crabapple Creek, but probably not by as much as the calculated values suggest, because Crabapple Creek has more individuals belonging to rare species which were not included in the analysis. Another way of looking at it is that a larger proportion of the total fauna of Bruce Creek was included in the productivity estimates. The reader should recall that these are not absolute annual productivity values and they are not corrected for differences in life history (cohort production intervals). They are meant only to provide a comparison of productivity over a specific time interval. It is unlikely that any of the productivity values for the species reported here would be

reduced by these corrections so that they may be thought of as minimum values. Ephemerella dorothea, a collector-gatherer mayfly was the dominant species (numbers and biomass) of Crabapple Creek. Lirceus fontinalis, an isopod shredder, was the dominant species in Bruce Creek. Differences in productivity of functional groups were more pronounced than differences in abundance of functional groups.

The productivity of shredders in Bruce Creek was found to be more than twice the productivity of shredders in Crabapple Creek. Accurate estimates of the export of POM from the streams were not obtained, but the export should equal the difference between input and the sum of standing stock and amount consumed (including the consumption by microorganisms). The input of CPOM to Bruce Creek was 1.7 times the input to Crabapple Creek but Bruce creek is a little smaller than Crabapple Creek so the density of CPOM entering Bruce Creek may in fact be closer to twice the density in Crabapple creek. The ratio of shredder productivity is similar to the ratio of CPOM input to the two streams. Increased transport of FPOM as a result of greater CPOM processing in Bruce Creek may partly explain the greater abundance and productivity of hydropsychid caddisflies in that stream.

Theoretical and Practical Implications

The rapidly developing body of theory in lotic ecology should prove useful in directing future study into the effects of surface mining coal on streams. A large number of studies have proven the usefulness and value of diversity and other indices of community structure in the detection and monitoring of environmental disturbance. However diver-

sity indices provide only partial insight into causal relationships between the disturbance and its effects on the community. The functional group approach advocated by Cummins (1974) and applied here demonstrates that streams can have relatively healthy looking diversity values but differ significantly in their functional organization.

The use of community structure has been and should continue to be an important tool in the analysis of lotic ecosystems. It is also an important prerequisite to a functional approach since many taxa must be identified to genus in order to correctly assign their functional group affiliation. Since the community structure of so many streams has been studied, it should be a relatively simple matter to go back and determine their functional group composition. When the functional group composition of more mined streams becomes available the significance of the findings from this dissertation can be evaluated. Diversity indices and other measures of community structure will continue to depend on the development and application of taxonomy but a functional group approach does not entirely circumvent this problem either since functional group classification of many taxa must be made at the generic level. Functional group classification depends in turn on an accurate understanding of the trophic ecology of lotic communities and this is also an incompletely defined subject. Many organisms undoubtedly have flexible diets or diets which change as they grow. A common pattern is a gradual shift towards predation as size increases. So, much work remains to be done in both the areas of community structure and function. It cannot be maintained that one approach is superior to the other since each has its merits. It can however, be stated uncategorically that community struc-

ture has been extensively studied, while community function is just beginning to be.

An interesting result of this study was the higher measured productivity of Bruce creek in spite of its having a lower total macroarthropod density. This result demonstrates the importance of not equating abundance with productivity. It also suggests that the large amount of functional redundancy in the fauna of streams may allow them to remain productive despite the reduction and/or loss of many taxa. The species which dominate the fauna of Bruce creek are probably the most opportunistic feeders. This is true of many of the shredders and of the collector-gatherer Epeorus dispar. It is possible that some of the species found in mined streams have altered their feeding. Many aquatic insects have been shown to do so. The great abundance of Lirceus fontinalis in Bruce Creek may partly reflect competitive release. Perhaps it is filling the feeding niche vacated by the collector-gatherers. If this were so then the functional group composition of the two streams differ mostly in periphyton grazers. Additional research into the trophic ecology of aquatic macroarthropods should strengthen the reliability of the functional group approach.

There are several areas of basic research particularly relevant to the effects of surface mining coal on lotic ecosystems which remain wanting. Carbon budgets have been ascertained for pristine and clear-cut streams but not surface mined streams. The results from this study are not sufficient to construct a complete carbon budget for these streams but the disproportionate ratio of input to standing stock between Crabapple and Bruce Creeks suggest their carbon budgets do

differ. Another potentially important area of research is the relationship between surface mining and the periphyton communities of streams. Results from this study indicate that a significant proportion of the macroarthropod community of first order streams is dependent on a healthy periphyton community. It is well known that microbial communities are important to many processes in stream ecosystems. The effect of surface mining coal on these organisms has never been adequately assessed. These three areas of research as well as the structural and functional properties of stream communities are all integral components of a comprehensive understanding of natural and degraded lotic ecosystems.

The resource spiralling concept and the river continuum concept should both prove useful in directing future research investigating the effects of surface mining on lotic ecosystems. This study was not intended to test the predictions of either theory but both provided useful paradigms from which ideas concerning the possible effects of surface mining on lotic ecosystems were formulated. However the potential for studying lotic ecosystems within the framework of these theories is great. Measuring the spiralling length of mined and unmined streams could yield considerable insight into the ecosystem level changes resulting from surface mining coal. The macroinvertebrate composition of Bruce Creek differed from Crabapple Creek in a manner consistent with predictions of the river continuum concept in that differences in the physical structure of a stream were reflected by the functional group composition of the fauna. However, the river continuum theory was formulated to explain differences in functional group organ-

ization of stream communities resulting from differences in the organic matter resources of streams that correlate with stream size (order).

The increased peak flows (assumed for Bruce Creek) did not seem to greatly affect its POM properties. Perhaps amounts of watershed disturbance more typical of surface mining in this area (10-30% of watershed area) would alter POM dynamics sufficiently to test this theory. However mined streams experience other effects, siltation and reduced periphyton productivity, which would complicate evaluation of the model. A more useful approach to applying the river continuum model would be to consider the effects of surface-mining coal as they are propagated downstream along the "river continuum." This approach has already been applied in healthy rivers.

The implications of this study for surface mine reclamation policy are not encouraging. Minear et al. (1980) have demonstrated that reclamation as currently practiced does not prevent tremendous amounts of siltation from occurring in streams. This siltation is probably the main factor affecting the collector-gatherers, which are primarily Ephemeropterans, and grazers, which are represented by several orders. Matter and Ney (1981) did not find much difference in the physical and biological characteristics of reclaimed versus unreclaimed streams. In addition the number of watersheds mined is constantly increasing and the cumulative effects on biological communities will undoubtedly also spread downstream. Virtually the entire watershed of some rivers, the Cumberland for example, contain mineable amounts of coal. Although some headwaters are protected, most are open to mining, and, as the amount of

watershed disturbance increases these rivers will undoubtedly experience increased taxonomic alteration and possible loss of many species.

LIST OF REFERENCES

LIST OF REFERENCES

Allen, K. R. 1949. Some aspects of the production and cropping of freshwaters. Trans. Roy. Soc. N.Z. 77:222-228.

Allen, K. R. 1951. The Horokiwi stream, a study of a trout population. N.Z. Mar. Dep. Fish. Bull. 10.

Allen, R. K. and G. F. Edmunds. 1963. A revision of the genus Ephemerella (Ephemeroptera:Ephemerellidae) VII. The subgenus Eurylophella in North America. Can. Entomol. 95:597-623.

Allen, R. K. and G. F. Edmunds. 1965. A revision of the genus Ephemerella (Ephemeroptera:Ephemerellidae) VIII. The subgenus Ephemerella in North America. Misc. Publ. Entomol. Soc. Amer. 4:244-282.

American Public Health Association (APHA). 1975. Standard Methods for the Examination of Water and Waste Water, 14th Edition. American Public Health Association, Washington, D.C. USA.

Anderson, R. O. and F. F. Hooper. 1956. Seasonal abundance and production of littoral bottom fauna in a southern Michigan lake. Trans. Amer. Microsc. Soc. 75:259-270

Anderson, N. H. and J. R. Sedell. 1979. Detritus processing by macroinvertebrates in streams. Ann. Rev. Entomol. 24:351-377.

Barlocher, F. and B. Kendrick 1975. Leaf-conditioning by microorganisms. Oecologia 20:359-362.

Bednarik, H. F. and W. P. MacCafferty. 1979. Biosystematic revision of the genus Stenonema (Ephemeroptera:Heptageniidae). Canadian Bull. Fish. Aquat. Sci. no. 201.

Benfield, E. F. and J. R. Webster. 1985. Shredder abundance and leaf breakdown in an Appalachian stream. Freshwater Biology 15:113-120.

Benke, A. C. and J. B. Wallace. 1980. Trophic basis of production amoung net-spinning Caddisflies in a southern Appalachian stream. Ecology 61:108-118.

Benke, A. C. 1984. Secondary production of aquatic insects. In Resh, V. H. and D. M. Rosenburg (eds.). The Ecology of Aquatic Insects. Praeger Publishing, New York.

Benke, A. C., T. C. Van Ardsdall, Jr., and D. M. Gillespie. 1984. Invertebrate productivity in a subtropical blackwater river: the importance of habitat and life history. Ecol. Monogr. 54:25-63

Bilby, R. E. and G. E. Likens. 1980. The importance of debris dams in the structure and function of stream ecosystems. Ecology 61:1107-1113.

Bilby, R. E. 1981. Role of organic debris dams in regulating the export of dissolved and particulate matter from a forested watershed. Ecology 62:1234-1243.

Boccardy, J. A. and W. M. Spaulding, Jr. 1968. Effects of surface mining on fish and wildlife in Appalachia. Bureau of Sport Fisheries Wildlife Resource Publ. no. 65. Washington, D.C.

Boling, R. H., Jr., E. D. Goodman, J. A. Van Sickle, J. O. Zimmer, K. W. Cummins, R. C. Petersen and S. R. Reice. 1975. Toward a model of detritus processing in a woodland stream. Ecology 56:141-151.

Boling, R. H., Jr., R. C. Petersen and K. W. Cummins. 1975. Ecosystem modeling for small woodland streams. In B. C. Patten (ed.). Systems Analysis and Simulation in Ecology, Vol. III. Academic Press, Inc., New York.

Boysen-Jensen, P. 1919. Valuation of the Limfjord. I. Studies on the fish-food in the Limfjord 1909-1917, its quantity, variation and annual production. Rep. Danish Biol. Sta. 26:3-44.

Branson, B. A. and D. L. Batch. 1972. Effects of stripminig on small-stream fishes in east-central Kentucky. Proc. Biol. Soc. Wash. 84:507-518.

Brigham, A. R. and D. D. Herlong. 1982. Aquatic and semiaquatic Lepidoptera. In Brigham, A. R., W. U. Brigham and A. Gnilka (eds.). Aquatic Insects and Oligochaetes of North and South Carolina. Midwest Aquatic Enterprises, Mahomet, Illinois

Brigham, A. R., W. U. Brigham and A. Gnilka, (eds.). 1982. Aquatic Insects and Oligochaetes of North and South Carolina. Midwest Aquatic Enterprises, Mahomet, Illinois

Brigham, W. U. 1982. Megaloptera. In Brigham, A. R., W. U. Brigham and A. Gnilka, (eds.). 1982. Aquatic Insects and Oligochaetes of North and South Carolina. Midwest Aquatic Enterprises, Mahomet, Illinois

Bright, G. R. 1982. Secondary benthic production in a tropical island stream. Limnol. Oceanogr. 27:472-480.

Brillouin. L. 1962. Science and Imformation Theory, 2nd ed. Academic Press. New York.

Cairns, J. and K. L. Dickson. 1971. A simple method for the biological assessment of the effects of waste discharge on aquatic bottom-dwelling organisms. J. Water Pollut. Control Fed. 43:755-772.

Carter, J. 1964. Stream bottom fauna. In Collier, C. R., R. J. Pickering and J. J. Musser (eds.). Influence of Stripmining on the Hydrologic

Environment of Parts of Beaver Creek Basin. U.S. Geol. Surv. Prof. Papers 427B.

Cherry, D. S., R. K. Guthrie, S. R. Larrick and F. F. Sherberger. 1979a. The influence of coal ash and thermal discharges on the distribution and bioaccumulation of aquatic invertebrates. Hydrobiologia 62:257-267.

Cherry, D. S., S. R. Larrick, R. K. Guthrie, E. M. Davis and F. F. Sherberger. 1979b. Recovery of vertebrate and invertebrate populations in a coal ash stressed drainage system. J. Fish. Res. Board Can. 36:1089-1096.

Chutter, F. M. and R. G. Noble. 1966. The reliability of a method of sampling stream invertebrates. Arch. Hydrobiol. 62:95-103.

Ciborowski, J. J. H., P. J. Pointing and L. D. Corkum. 1977. The effect of current velocity and sediment on the drift of the mayfly Ephemerella subvaria McDunnough. Freshwater Biology 7:567-572.

Comiskey, C. E. 1978. Aspects of the organic carbon cycle on Walker Branch watershed: a study of land water/interaction. Dissertation. University of Tennessee, Knoxville, Tennessee, USA.

Cummins, K. W. 1973. Trophic relations of aquatic insects. Ann. Rev. Entomol. 18:183-206.

Cummins, K. W. 1974. Structure and Function of stream ecosystems. Bioscience 24:631-641.

Cummins, K. W. and M. J. Klug. 1979. Feeding ecology of stream invertebrates. Ann. Rev. Ecol. Systemat. 10:147-172.

Cummins, K. W., G. L. Spengler, G. M. Ward, R. M. Speaker, R. W. Ovink, D. C. Mahan and R. L. Mattingly. 1980. Processing of confined and naturally entrained leaf litter in a woodland stream ecosystem. Limnol. and Oceanogr. 25:952-957.

Curtis, W. R. 1972a. Chemical changes in streamflow following surface mining in eastern Kentucky. Fourth Symposium on Coal Mine Drainage Research. Mellon Institute, Pittsburg, Pa.

Curtis, W. R. 1972b. Stripmining increases flood potential of mountain watersheds. Proc. Nat. Symp. on Watersheds in Transition. Am. Water Res. Assoc. and Colorado State University.

Curtis, W. R. 1977. Hydrologic aspects of surface mining in the east. Proc. of the Soc. American Foresters.

Cushman, R. M., J. W. Elwood and S. G. Hildebrand. 1975. Production dynamics of Alloperla mediana Banks (Plecoptera:Chloroperlidae) and

Diplectrona modesta (Trichoptera:Hydropsychidae) in Walker Branch, Tennessee. Environmental Sciences Division Publication 785, Oak Ridge National Laboratory, Oak Ridge, Tennessee, USA.

Cushman, R. W., J. W. Elwood and S. G. Hildebrand. 1977. Life History and production of Alloperla mediana and Diplectrona modesta in Walker Branch, Tennessee. Amer. Mid. Nat. 98:354-364.

Cushman, R. W., H. H. Shugart, Jr., S. G. Hildebrand and J. W. Elwood. 1978. The effect of growth curve and sampling regime on instantaneous-growth, removal-summation, and Hynes/Hamilton estimates of aquatic insect production: a computer simulation. Limnol. Oceanogr. 23:184-189.

de la Cruz, A. A., and H. A. Post. 1977. Production and transport of organic matter in a woodland stream. Arch. Hydrobiol. 80:227-238.

De March, B. G. E. 1976. Spatial and temporal patterns in macrobenthic stream diversity. J. Fish. Res. Board Can. 33:1261-1270.

Dyer, K. L., and W. R. Curtis. 1977. Effects of stripmining on water quality in small streams in eastern Kentucky, 1967-1975: U.S. Department of Agriculture, Forest Service Researh Paper NE-372.

Dyer, K. L. 1982. Stream water quality in the coal region of Tennessee. Broomall, Pa.: Northeast For. Exp. Stn.; USDA For. Serv. Gen. Tech. Rep. NE-77.

Edmunds, G. F., Jr., S. L. Jensen and L. Berner. 1979. The Mayflies of North and Central America. University of Minnesota Press, Minneapolis.

Elwood, J. W. and D. J. Nelson. 1972. Periphyton production and grazing rates in a stream measured with ^{32}P material balance method. Oikos 23:295-303.

Elwood, J. W. and J. D. Newbold, A. F. Trimble and R. W. Stark. 1981. The limiting role of Phosphorus in a woodland stream ecosystem: effects of P enrichment on leaf decomposition and primary producers. Ecology 62:146-158.

Elwood, J. W., J. D. Newbold, R. V. O'Neill and W. Van Winkle. 1983. Resource spiralling: an operational paradigm for analysing lotic ecosystems, In T. D. Fontaine, III, and S. M. Bartell (eds.). The Dynamics of Lotic Ecosystems. Ann Arbor Press, Ann Arbor, MI.

Etnier, D. A. and G. A. Shuster. 1979. An annotated list of the Trichoptera (caddisflies) of Tennessee. J. Tennessee Acad. Sci. 54:15-22.

Fairchild, J. F., T. P. Boyle, E. Robinson-Wilson and J. R. Jones. 1983. Microbial action in detrital leaf processing and the effects of chemical perturbation. In T. D. Fontaine, III, and S. M. Bartell (eds.). Dynamics of Lotic Ecosystems. Ann Arbor Science, Ann Arbor, MI.

Fenneman, N. M. 1938. Physiography of the Eastern United States. Mcgraw-Hill, New York.

Fisher, S. G. and G. E. Likens. 1973. Energy flow in Bear Brook, New Hampshire: An integrative approach to stream ecosystem metabolism. Ecol. Monogr. 43:421-439.

Freeman, M. C. and J. B. Wallace. 1984. Production of net-spinning caddisflies (Hydropsychidae) and black flies (Simuliidae) on rock outcrop substrate in a small southeastern piedmont stream. Hydrobiologia 112:3-5.

Freund, R. J. and R. C. Littell. 1981. SAS for Linear Models, A Guide to the ANOVA and GLM Procedures. SAS Institute Inc. Cary, NC.

Frison, T. H. 1942. Studies of North American Plecoptera fauna with special reference to the fauna of Illinois. Illinois Nat. Hist. Surv. Bull. 22:232-355.

Garman, G. C. and T. F. Waters. 1983. Use of the size-frequency (Hynes) method to estimate annual production of a stream fish population. Can. J. Fish. Aquat. Sci. 40:2030-2034.

Gaufin, A. R., E. K. Harris and H. J. Walter. 1956. A statistical evaluation of stream bottom sampling data obtained from three standard samplers. Ecology 36:643-648.

Green, R. H. 1979. Sampling Design and Statistical Methods for Environmental Biologists. John Wiley and Sons, New York.

Hamilton, A. L. 1969. On estimating annual Production. Limnol. Oceanogr. 25:773-777.

Harper, P. P. and H. B. N. Hynes. 1971. The Leuctridae of eastern Canada. (Insecta; Plecoptera). Can. J. Zool. 49:915-920.

Hawkins, C. P. and J. R. Sedell. 1981. Longitudinal and seasonal changes in functional organization of macroinvertebrate communities in four Oregon streams. Ecology 62:387-397.

Hensley, J. P. 1970. Stream bottom fauna. In Collier, C. R., R. J. Pickering and J. J. Musser (eds.). Influence of Stripmining on the Hydrologic Environment of Parts of Beaver Creek Basin. U.S. Geol. Surv. Prof. Papers 427C.

Herricks, E. E. and J. Cairns. 1974. Rehabilitation of Streams Receiving Acid Mine Drainage. Virginia Water Resources Research Center, Bulletin #66. VPI-SU, Blacksburg, Va.

Hitchcock, S. W. 1974. Guide to the Insects of Connecticut. Part VII. The Plecoptera or Stoneflies of Connecticut. Connecticut State Geol. and Nat. Hist. Sur. Bull. No. 107.

Huggins,D. G. and W. U. Brigham. 1982. Odonata. In Brigham, A. R., W. U. Brigham and A. Gnilka (eds.). Aquatic Insects and Oligochaetes of North and South Carolina. Midwest Aquatic Enterprises, Mahomet, Illinois

Hubricht, L. 1949. The Freshwater Isopods of the Genus Lirceus (Asellota, Asellidae). Amer. Mid. Nat. 42:334-349.

Hynes, H. B. N. 1961. The invertebrate fauna of a Welsh mountain stream. Arch. Hydrobiol. 57:344-388.

Hynes, H. B. N. and M. J. Coleman. 1968. A simple method of assessing the annual production of stream benthos. Limnol. Oceanogr. 13:569-573.

Hynes, H. B. N. 1961. Groundwater and stream ecology. Hydrobiologia 100:93-99.

Jackson, D., C. F. Mason, and S. P. Long. 1985. Macro-invertebrate populations and production on a salt-marsh in east England dominated by Spartina Anglica. Oecologia 65:406-411.

Jop, K. and S. W. Sxczytko. 1984. Life cycle and production of Isoperla signata (Banks) in a central Wisconsin trout stream. Aquat. Ins. 6:81-100.

Kaushik, N. K. and H. B. N. Hynes. 1971. The fate of leaves that fall into streams. Arch. Hydrobiol. 68:465-515.

Kostalos, M. and L. R. Seymour. 1976. Role of microbial enriched detritus in the nutrition of Gammarus minus. Oikos 28:512-516.

Kovalak, W. P. 1981. Assessment and Prediction of impacts of effluents on communities of benthic stream macroinvertebrates, In M. Bates and C. I. Weber (eds.). Ecological Assessments of Effluent Impacts on Communities of Indigenous Aquatic Organisms, ASTM Special Technical Report 730. ASTM, Philadelphia, PA.

Krueger, C. C. and F. B. Martin. 1980. A note on the Hynes method of estimating secondary production. Limnol. Oceanogr. 25:770-773.

Ladle, M., J. S. Welton, and J. A. B. Bass. 1984. Larval growth and production of three species of Chironomidae from an experimental recirculating stream. Arch. Hydrobiol. 102:201-214.

Lamberti, G. A. and V. H. Resh. 1983. Stream periphyton and insect herbivores: an experimental study of grazing by a caddisfly population. Ecology 65:1124-35.

Larrick, S. R., J. R. Clark, D. S. Cherry and J. Cairns. 1981. Structural and functional changes of aquatic heterotrophic bacteria to thermal, heavy, and fly ash effluents. Water Research 15:875-880.

Leist, D. W., F. Quinones, D. S. Mull, and M. Young. 1982. Hydrology of Area 15, Eastern Coal Province, Kentucky and Tennessee: U.S. Geological Survey Open-file Report 81-809.

Leopold, L., M. G. Wolman, J. P. Miller. 1964. Fluvial Processes in Geomorphology. W. H. Freeman, San Francisco.

Lowery, J. F., P. H. Counts, H. L. Edmiston and F. D. Edwards. 1983. Water Resources Data for Tennessee, 1983. U.S. Geological Survey Water-data Report TN 78-1.

Lush, D. L. and H. B. N. Hynes. 1973. The Formation of particles in freshwater leachates of dead leaves. Limnol. Oceanogr. 18:968-977.

Martien, R. F. and A. C. Benke. 1977. Distribution and production of two crustaceans in a wetland pond. Amer. Midl. Nat. 98:162-175.

Matter, W. J. 1978. The Status and seasonal dynamics of fish and benthic invertebrate populations in relation to organic and inorganic material inputs and surface mining impacts in three Virginia headwater streams. Dissertation. Virginia Polytech. Inst. and State Univ., Blacksburg, Va.

Matter, W. J. and J. J. Ney. 1981. The impact of surface mine reclamation on headwater streams in southwest Virginia. Hydrobiol. 78:63-71.

McClure, R. G. and K. W. Stewart. 1976. Life cycle and production of the mayfly Choroterpes (Neochoroterpes) mexicanus Allen (Ephemeroptera: Leptophlebiidae). Ann. Entomol. Soc. Amer. 69:134-144.

Mcdowell, W. H. and S. G. Fisher. 1976. Autumnal processing of dissolved organic matter in a small woodland stream ecosystem. Ecology 57:561-569.

McIntire, C. D. 1973. Periphyton dynamics in laboratory streams: a simulation model and its implications. Ecol. Monogr. 43:399-420.

Merritt, R. W. and K. W. Cummins. 1984. An Introduction to the Aquatic Insects of North America. Kendall/Hunt Publishing Co., Dubuque, IA.

Minear, R. A., B. A. Tschantz, J. H. Rule, G. L. Vaughan, D. E. Overton, and G. Briggs. 1976. Environmental aspects of coal production in the Appalachian Region. Appalachian Resources Project Report. Prep. for USDOE unfer contract # E-(40-1)-4946. Univ. of Tenn. Env. Center, Knoxville, Tenn.

Minear, R. A., B. A. Tschantz, J. H. Rule, G. L. Vaughan, D. E. Overton, and G. Briggs. 1977. Environmental aspects of coal production in the Appalachian Region. Appalachian Resources Project Report. Prep. for USDOE unfer contract # E-(40-1)-4946. Univ. of Tenn. Env. Center, Knoxville, Tenn.

Minear, R. A., B. A. Tschantz, J. H. Rule, G. L. Vaughan, D. E. Overton, and G. Briggs. 1978. Environmental aspects of coal production in the Appalachian Region. Appalachian Resources Project Report. Prep. for USDOE unfer contract # EY-76-S-05-4946. Univ. of Tenn. Env. Center, Knoxville, Tenn.

Minear, R. A., D. E. Overton, G. L. Vaughan and B. A. Tschantz. 1979. Environmental aspects of coal production in the Appalachian Region. Appalachian Resources Project Report. Prep. for USDOE unfer contract # EY-76-S-05-4946. Univ. of Tenn. Env. Center, Knoxville, Tenn.

Minear, R. A., D. E. Overton, G. L. Vaughan and B. A. Tschantz. 1980. Environmental aspects of coal production in the Appalachian Region. Appalachian Resources Project Report. Prep. for USDOE unfer contract # EY-76-S-05-4946. Univ. of Tenn. Env. Center, Knoxville, Tenn.

Minshall, J. N. 1967. Life history and ecology of Eeorus pleuralis (Banks) (Ephemeroptera:Heptageniidae). Amer. Midl. Nat. 78:369-388.

Minshall, G. W., R. C. Petersen, K. W. Cummins, T. L. Bott, J.R. Sedell, C. E. Cushing and R. L. Vannote. 1983. Interbiome comparison of stream ecosystem dynamics. Ecol. Monogr. 53:1-25.

Needham, P. R. and R. L. Usinger. 1956. Variability in the macrofauna of a single riffle in Prospect Creek, California, as indicated by the Surber sampler. Hilgardia 24:383-409.

Newbold, J. D., D. C. Erman and K. B. Roby. 1980. Effects of logging on macroinvertebrates in streams with and without buffer strips. Can. J. Fish. Aquat. Sci. 37:1076-1085.

Newbold, J. D., J. W. Elwood, R. V. O'Neill and W. Van Winkle. 1981. Measuring nutrient spiralling in streams. Can. J. Fish. and Aquat. Sci. 38:860-863.

Newbold, J. D., P. J. Mulholland, J. W. Elwood, and R. V. O'Neill. 1982. Organic carbon spiralling in stream ecosystems. Oikos 38:266-272.

Newman, R. M. and F. B. Martin. 1983. Estimation of fish production rates and associated variances. Can. J. Fish. Aquat. Sci. 40:1729-1736.

O'Hop, J., J. B. Wallace and J. D. Haefner. 1984. Production of a stream shredder, Peltoperla maria (Plecoptera:Peltoperlidae), in disturbed and undisturbed hardwood catchments. Freshwater Biology 14: 13-21.

Pask, W. M. and R. M. Costa. 1971. Efficiency of sucrose floatation in recovering insect larvae from benthic stream samples. Can. Entomol. 103:1649-1652.

Petersen, R. C. and K. W. Cummins. 1974. Leaf processing in a woodland stream. Freshwater Biology 4:343-368.

Pielou, E. C. 1975. Ecological Diversity. John Wiley and Sons, New York.

Rabeni, C. F. and G. W. Minshall. 1977. Factors affecting microdistribution of stream benthic insects. Oikos 29:33-43.

Reice, S. R. 1974. Environmental patchiness and the breakdown of leaf litter in a woodland stream. Ecology 55:1271-1282.

Resh, V. H. 1977. Habitat and substrate influences on population and production of a stream caddisfly, Ceraclea ancylus (Leptoceridae). Freshwater Biology 7:261-278.

Resh, V. H. 1979. Sampling variability and life history features: basic considerations in the design of aquatic insect studies. J. Fish. Res. Board Can. 36:290-311.

Ricker, W. E. 1946. Production and utilization of fish populations. Ecol. Monogr. 16:373-391.

Roback, S. S. and J. W. Richardson. 1969. The effects of acid mine drainage on aquatic insects. Proc. Acad. Nat. Sci. Philadelphia 121:81-107.

Rogers, L. E. , W. T. Hinds and R. L. Buschbom. 1976. A general weight vs. length relationship for insects. Ann. Entomol. Soc. Amer. 69:387-389.

Sanders, H. L. 1956. The biology of marine bottom communities. Bull. Bingham Oceonogr. Collect. 15:345-414.

SAS Institute Inc. 1982. Sas User's Guide: Basics, 1982 Edition. SAS Institute Inc., Cary, NC.

Schuster, G. G. and D. A. Etnier. 1978. A Manual for the Identification of the Larvae of the Caddisfly Genera Hydropsyche Pictet and Symphitopsyche Ulmer in Eastern and Central North America (Trichoptera:Hydropsychidae). EPA-60014-78-060.

Sedell, J. R., F. J. Triska, J. D. Hall, N. H. Anderson and J. H. Lyford. 1974. Sources and fates of organic inputs in coniferous forest streams. In Waring, R. H. and R. L. Edmunds (eds.). Integrated Research in the Coniferous Forest Biome, Conif. For. Biome, Ecosyst. Anal. Stud., US/IBP, Bull. No 5, University of Washington, Seattle.

Shannon, C. E. and W. Weaver. 1949. The Mathematical Theory of Communication. University of Illinois Press, Urbana.

Short, A. S. and P. E. Maslin. 1977. Processing of leaf litter by a stream detrivore: effect on nutrient availabilty to collectors. Ecology 58:935-938.

Simpson, E. H. 1949. Measurement of diversity. Nature 163:688.

Smock, L. A. 1980. Relationshps between body size and biomass of aquatic insects. Freshwater Biology 10:375-383.

Snedecor, G. W. and W. G. Cochran. 1967. Statistical Methods. The Iowa State University Press. Ames, Iowa.

Specht, W. L., D. S. Cherry, R. A. Lechleitner and J. Cairns. 1984. Structural, functional, and recovery responses of stream invertebrates to fly ash effluent. Can. J. Fish. Aquat. Sci. 41:884-896.

Stair, D. M. and V. R. Tolbert. 1980. Response of benthic insect species to changes in stream velocity resulting from stripmining disturbance. Symposium proceedings. North American Benthological Society, Special Symposia: Applied Periphyton, Estuarine Oligochaetes, Mining Effects on Benthos.

Suberkropp, K. and M. J. Klug. 1976. Fungi and bacteria associated with leaves during processing in a woodland stream. Ecology 57:707-719.

Talak, A. 1977. The recovery of stream benthic insect communities following coal stripmining in the Cumberland mountains of Tennessee. Thesis. University of Tennessee, Knoxville.

Teal, J. M. 1957. Community metabolism in a temperate cold spring. Ecol. Monogr. 27:283-302.

Tolbert, V. R. 1978. Effects of stripmining related disturbance on benthic insect communities of selected streams in the New River

basin of east Tennessee. Dissertation. University of Tennessee, Knoxville.

Triska, F. J., J. R. Sedell, K. Cromack, Jr., S. V. Gregory and F. M. Mc Corison. 1984. Nitrogen budget for a small coniferous forest stream. Ecolog. Monogr. 54:119-140.

Unzicker, J. D. and P. H. Carlson. 1982. Ephemeroptera. In Brigham, A. R., W. U. Brigham and A. Gnilka (eds.). Aquatic Insects and Oligochaetes of North and South Carolina. Midwest Aquatic Enterprises, Mahomet, Illinois

Unzicker, J. D. and Von H. McCaskill. 1982. Plecoptera. In Brigham, A. R., W. U. Brigham and A. Gnilka (eds.). Aquatic Insects and Oligochaetes of North and South Carolina. Midwest Aquatic Enterprises, Mahomet, Illinois

Unzicker, J. D., V. H. Resh and J. C. Morse. 1982. Trichoptera. In Brigham, A. R., W. U. Brigham and A. Gnilka (eds.). Aquatic Insects and Oligochaetes of North and South Carolina. Midwest Aquatic Enterprises, Mahomet, Illinois

U.S. Geological Survey. 1983. Water Resources Data for Tennessee Annual Reports. U.S. Geological Survey, Nashville TN.

U.S. Soil Conservation Service. 1979. The Status of Land Disturbed by Surface Mining in the United States. Basic Statistics by State and County as of July 1, 1977. SCS-TP-158.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell and C. E. Cushing. 1980. The river continuum concept. Can. J. Fish. Aquat. Sci. 37:130-137.

Vaughan, G. L., A. Talak and R. J. Anderson. 1978. The chronology and character of recovery of aquatic communities from the effects of strip mining for coal in east Tennessee. Symposium proceedings. Surface Mining and Fish/Wildlife needs in the eastern United States. West Virginia University and the U.S. Department of the Interior, Fish and Wildlife Service.

Vaughan, G. L. 1979. Effects of stripminig on fish and diatoms in streams of the New River drainage basin. J. Tenn. Acad. Sci. 54:110-114.

Vaughan, G. L. 1980. Rapid recovery of aquatic communities in Cumberland Mountain streams following contour mining for coal. Symposium proceedings. North American Benthological Society, Special Symposia: Applied Periphyton, Estuarine Oligochaetes, Mining Effects on Benthos.

Vaughan, G. L., L. Minter and J. Schiller. 1982. New River Project data bases and documentation, joint research. Volume 2. Biological and associated water quality data. Prepared by the University of Tennessse, Knoxville, Departments of Civil Engineering and Zoology under contract # DOE-AS05-76EV094946(141080-6237-R95; ERDA E(40-1)4946).

Waters, T. F. and G. W. Crawford. 1973. Annual production of a stream mayfly population: a comparison of methods. Limnol. Oceanogr. 18:286-296.

Waters, T. F. 1977. Secondary production in inland waters. In MacFayden, A. (ed.). Advances in Ecological Research, Volume 10. Academic Press, New York.

Waters, T. F. 1979. Influence of benthos life history upon the estimation of secondary production. J. Fish. Res. Board Can. 36:1425-1430.

Webb, D. W. and W. U. Brigham. 1982. Aquatic Coleoptera. In Brigham, A. R., W. U. Brigham and A. Gnilka (eds.). Aquatic Insects and Oligochaetes of North and South Carolina. Midwest Aquatic Enterprises, Mahomet, Illinois

Weber, C. I. 1973. Biological Field and Laboratory Methods for Measuring the Quality of Surface Waters and Effluents. U.S. Environmental Protection Agency, EPA-670/4-73-001.

Webster, J. R. 1977. Large particulate organic matter processing in stream ecosystems. In Corell, D. L. (ed.) Watershed Research in Eastern North America. Smithsonian Institution, Washington, D.C.

Webster, J. R. and B. C. Patten. 1979. Effects of watershed perturbation on stream potassium and calcium budgets. Ecol. Monogr. 49:51-72.

Webster, J. R. and J. B. Waide. 1982. Effects of clearcutting on leaf breakdown in a southern Appalachian stream. Freshwater Biology 12:331-334.

Webster, J. R. 1983. The role of benthic macroinvertebrates in detritus dynamics of streams: a computer simulation. Ecol. Monogr. 53:383-404.

Wetzel, R. G. and B. A. Manny. 1977. Seasonal changes in particulate and dissolved organic carbon and nitrogen in a hardwater stream. Arch. Hydrobiol. 80:29-39.

Whitford, L. A. and G. J. Shumacher. 1961. Effect of current on mineral uptake and respiration by a freshwater alga. Limnol. Oceanogr. 6:423-725.

Wiederholm, T. 1984. Responses of aquatic insects to environmental pollution. In Resh, V. H. and D. M. Rosenberg (eds.). The Ecology of Aquatic Insects. Praeger Publishers, New York.

Wiggins, G. B. 1978. Larvae of the North American Caddisfly Genera (Trichoptera). University of Toronto Press, Toronto, Canada.

Wilhm, J. L. and T. C. Dorris. 1968. Biological parameters for water quality criteria. Bioscience 18:477-481.

Williams, E. B. 1981. Benthic insect community structure in Cumberland mountain streams twenty-five years after coal strip mining. Thesis. University of Tennessee, Knoxville, TN.

APPENDICES

APPENDIX A

Table A1. Water quality parameters for Crabapple and Bruce Creeks.

	Crabapple						Bruce					
	pH	Alk	Hrd	SO_4	Fe	Trb	pH	Alk	Hrd	SO_4	Fe	Trb
Date		mg/l				ftu		mg/l				ftu
121281	ns	20	20	0.0	0.00	1	ns	20	70	48.0	0.00	4
32182	ns	20	20	0.0	0.00	2	ns	20	55	55.0	0.00	50
41082	6.5	20	30	0.0	0.00	3	6.5	20	80	49.0	0.00	7
42482	6.7	20	30	0.0	0.00	3	7.3	20	90	57.5	0.00	8
50182	6.8	20	20	0.0	0.00	5	7.2	20	70	42.5	0.00	20
52382	6.8	20	20	0.0	0.10	13	7.2	20	80	48.5	0.00	88
53082	7.6	20	20	0.0	0.00	0	7.5	20	60	31.0	0.00	20
61982	7.0	20	20	0.0	0.00	3	7.1	20	110	80.0	0.00	14
71882	ns	20	20	0.0	0.10	17	7.3	20	130	80.0	0.00	35
81582	6.5	20	20	0.0	0.00	5	6.9	20	110	75.0	0.00	60
91282	7.7	20	20	0.0	0.00	2	7.4	20	110	72.5	0.00	7
111482	6.9	20	20	0.0	0.00	0	6.9	20	90	49.5	0.00	6
121882	7.5	20	20	0.0	0.17	1	7.7	20	60	42.0	0.12	9
12383	7.4	20	20	0.0	0.04	3	7.6	20	60	55.0	0.00	4
21983	7.4	20	20	0.0	0.00	5	7.4	15	80	70.0	0.25	13
31683	7.7	20	20	ns	0.95	7	7.4	20	90	ns	1.37	15
41783	6.7	20	20	0.0	0.17	6	6.8	25	65	53.0	0.60	12
52383	7.0	10	20	0.0	0.00	0	6.5	10	80	71.5	0.20	10
10784	7.1	20	20	6.0	0.10	1	6.6	20	60	50.5	0.10	3
20484	7.7	20	20	0.0	0.15	1	7.4	20	80	57.5	0.10	4
30384	7.9	20	20	0.0	0.25	3	6.9	20	40	52.5	3.20	33
33184	6.8	20	20	0.0	0.20	3	6.4	20	80	56.5	0.12	14
50584	6.4	20	20	7.5	0.20	4	6.2	20	50	35.0	0.10	25
52684	6.1	20	20	5.0	0.17	4	7.0	20	130	77.5	0.10	13
Mean	7.1	20	21	0.8	0.11	4	7.1	20	80	56.9	0.26	20

ns = no sample

Table A2. POM collected in fall-in sacks along Crabapple and Bruce Creeks from March 1982 to May 1983.

	Crabapple Creek				Bruce Creek			
Date	Mean CPOM	Mean FPOM	Mean wood	Sample size	Mean CPOM	Mean FPOM	Mean wood	Sample size
	g/m^2				g/m^2			
32782	5.68	0.83	10.36	16	7.52	0.72	7.21	16
41782	5.94	0.62	23.71	16	9.59	1.33	39.59	16
51582	7.37	1.06	1.75	16	8.72	1.01	1.17	16
61382	8.48	1.33	2.67	16	11.45	1.16	2.45	16
71082	4.41	0.95	0.69	16	5.48	0.61	0.77	16
80782	1.34	8.13	1.02	16	1.42	8.26	0.68	16
90482	27.71	2.73	0.26	16	28.56	1.86	15.53	15
102482	242.37	4.66	2.97	16	235.96	4.27	0.76	14
112182	64.97	1.16	1.61	16	73.57	1.68	3.36	16
11583	4.98	0.37	18.39	16	4.96	0.53	8.38	14
21383	1.59	0.22	1.82	15	0.90	0.17	0.36	14
41083	2.15	0.42	1.67	14	1.09	0.02	2.60	13
51483	8.34	0.26	3.78	16	8.90	0.51	5.46	15
totals	385.35	22.74	70.69		398.13	22.13	88.34	

Table A3. POM collected in blow-in nets along Crabapple and Bruce Creeks from April 1982 to May 1983.

	Crabapple Creek			Bruce Creek		
Date	Mean CPOM g/m	Mean wood g/m	Sample size	Mean CPOM g/m	Mean wood g/m	Sample size
			bench			
41782	5.23	1.83	12	19.66	1.57	7
51582	0.55	0.79	11	0.47	0.30	12
61382	0.17	0.06	12	0.10	0.10	8
71082	0.17	0.06	12	0.19	0.41	9
80782	0.15	0.10	12	0.50	0.11	11
90482	0.84	0.16	12	1.19	0.33	12
102482	14.78	0.70	9	22.40	0.05	8
112182	9.18	0.05	10	40.18	0.88	12
11583	3.78	0.09	8	16.94	0.87	12
21383	7.08	0.60	10	7.96	0.51	12
31383	2.80	0.94	11	10.20	0.31	10
41083	5.41	0.41	11	12.28	0.81	11
51483	3.25	0.45	12	13.03	0.62	12
totals	53.40	6.24		145.10	6.87	
			slope			
41782	67.73	11.27	11	58.34	27.35	11
51582	10.04	2.74	11	6.47	1.19	12
61382	2.15	1.10	12	4.22	0.39	11
71082	2.79	1.38	11	1.32	1.92	11
80782	3.17	0.92	11	1.44	0.78	11
90482	5.60	1.21	11	4.34	6.79	10
102482	29.23	2.48	10	66.04	2.46	11
112182	43.31	2.51	12	66.87	3.56	12
11583	14.16	1.14	11	41.76	3.87	12
21383	16.28	3.13	7	11.35	1.82	12
31383	17.50	1.25	11	16.42	1.24	11
41083	34.43	1.93	12	23.21	3.11	12
51483	18.86	5.88	11	26.15	3.81	12
totals	318.64	43.16		473.02	65.15	

Table A4. POM collected by surber samplers in riffle areas of Crabapple and Bruce Creeks.

	Crabapple Creek				Bruce Creek			
Date	Mean cpom	Mean fpom	Mean wood	Sample size	Mean cpom	Mean fpom	Mean wood	Sample size
	(g/m^2)				(g/m^2)			
12383	26.89	4.69	10.71	16	28.00	1.81	11.91	14
31683	16.90	5.21	13.18	16	13.80	2.22	13.59	15
41783	14.07	3.25	11.87	16	10.80	1.96	15.22	16
10784	21.52	3.28	9.47	16	26.81	2.04	12.67	16
20484	16.61	2.96	18.40	16	22.01	2.39	19.97	16
30384	37.70	9.21	32.80	14	13.97	2.24	8.99	16
33184	14.51	3.70	8.59	16	23.69	3.60	29.67	16
50584	47.34	4.69	7.77	16	15.79	2.97	13.57	16
52684	54.34	2.47	7.49	16	8.37	1.33	9.67	16
totals	249.88	39.46	120.28		163.24	20.55	135.29	
means	27.76	4.38	13.36		18.14	2.28	15.03	

Table A5. POM collected by drift nets in riffle areas of Crabapple and Bruce Creeks.

	Crabapple Creek			Bruce Creek		
	CPOM	FPOM	wood	CPOM	FPOM	wood
Date	g/hr			g/hr		
41082	ns	ns	ns	0.45	.	0.38
41082	ns	ns	ns	0.80	0.45	0.90
53082	0.39	1.76	0.00	ns	ns	ns
61982	0.82	0.18	0.58	0.03	0.00	0.00
61982	0.11	0.00	0.02	0.03	0.00	0.00
71182	0.03	0.02	0.00	0.05	0.02	0.06
71182	0.18	0.13	0.01	0.03	0.03	0.01
91282	0.33	0.08	0.00	0.22	0.01	0.10
111482	0.02	0.01	0.00	0.17	0.01	0.00
111482	0.28	0.01	0.00	0.33	0.01	0.26
12383	1.76	0.38	0.08	0.25	0.08	0.27
12383	2.54	0.36	0.33	0.54	0.13	0.32
21383	0.20	0.09	0.00	ns	ns	ns
31683	0.32	0.06	0.00	0.12	0.03	0.00
41783	0.34	0.12	0.15	0.32	0.14	0.19
41783	0.62	0.09	0.10	0.58	0.15	0.16
10784	0.13	0.00	0.00	0.02	0.00	0.00
10784	0.16	0.04	0.05	0.43	0.03	0.11
20484	0.23	0.04	0.03	0.33	0.01	0.11
20484	0.06	0.02	0.02	0.83	0.01	0.43
30384	0.11	0.03	0.00	0.16	0.02	.
30384	0.41	0.06	0.15	0.13	0.02	0.01
33184	2.00	0.64	0.18	1.99	0.98	0.52
33184	0.89	0.41	0.11	2.12	0.67	0.19
50584	1.95	0.68	0.36	3.79	1.55	0.30
50584	6.01	4.14	0.94	11.25	4.47	1.09
52684	0.13	0.03	0.03	0.11	0.04	0.05
52684	0.07	0.01	0.01	0.11	0.02	0.00
mean	0.77	0.36	0.12	0.68	0.36	0.22

ns = no sample

Table A6. Total and volatile suspended solids in Crabapple and Bruce Creeks.

	Crabapple Creek			Bruce Creek		
	Total sus sol	Volatile sus sol		Total ss sus sol	Volatile sus sol	
Date	mg/l		Sample size	mg/l		Sample size
41082	4.00	6.00	2	16.00	2.00	2
42482	6.00	4.00	4	19.00	4.00	4
50182	4.67	0.67	6	24.00	2.67	6
52382	10.67	3.33	6	108.00	12.00	5
53082	10.63	4.00	6	49.17	6.50	6
61982	5.17	1.17	6	42.67	5.42	6
71882	7.68	2.64	5	41.47	5.00	6
81582	8.20	6.20	6	23.52	20.00	5
91282	5.60	1.80	6	20.00	2.80	4
111482	1.47	1.47	6	28.80	4.87	6
121882	5.87	1.60	6	10.72	2.40	5
21983	5.12	3.20	5	21.60	5.87	6
52583	10.00	2.80	2	28.60	5.00	6
10784	2.80	1.60	6	5.13	1.80	6
30384	2.27	0.80	3	9.40	1.27	6
33184	4.00	2.53	6	16.33	3.27	6
50684	8.40	2.93	6	36.87	4.60	6
52684	1.20	1.13	6	8.96	2.24	5
mean	5.78	2.66		28.35	5.10	

APPENDIX B

Table B1. Taxonomic checklist for ceratopogonids, chironomids, and miscellaneous dipterans in Crabapple and Bruce Creeks

Crabapple Creek	Bruce Creek
Atrichopogon sp.	Bezzia/Palpomyia sp.
Bezzia/Palpomyia sp.	Cardiocladius obscurus
Cardiocladius obscurus	Cladotanytarsus sp.
Cladotanytarsus sp.	Conchapelopia sp.
Conchapelopia sp.	Cricotopus sp.
Cricotopus annulator	Cricotopus tremulus
Cricotopus tremulus	Eukiefferiella brevicalcar
Cryptochironomus fulvus	Eukiefferiella rectangularis
Eukiefferiella brevicalcar	Eukiefferiella sp. a
Eukiefferiella rectangularis	Heleniella sp.
Eukiefferiella sp. a	Heterotrissocladius marcidus
Heleniella sp.	Nanocladius sp.
Heterotrissocladius marcidus	Parachaetocladius sp.
Krenosmittia sp.	Parametriocnemus lundbecki
Larsia sp.	Polypedilum illinoense
Micropsectra sp.	Rheotanytarsus sp.
Microtendipes sp.	Stempellina sp.
Parachaetocladius sp.	Stictochironomus devinctus
Parametriocnemus lundbecki	Tanytarsus sp.
Pericoma sp.	Tvetenia discoloripes
Polypedilum convictum	Empididae sp.
Polypedilum illinoense	
Rheocricotopus sp.	
Rheosmittia sp.	
Rheotanytarsus sp.	
Stempellina montivaga	
Stempellina sp.	
Tanytarsus sp.	
Thienemanniella xena	
Tvetenia bavarica	
Tvetenia discoloripes	
Zavrelia sp.	
Diamesinae sp.	
Empididae sp.	
Chrysops sp.	
Ormosa sp.	

Table B2. Functional group classification (Merritt and Cummins 1984).

Taxa	Functional group
Coleoptera	
Elmidae	
Ukn	scraper
Eubriidae	
Ectopria nervosa	scraper
Hydrophilidae	
Ukn	predator
Psephenidae	
Psephenus herricki	scraper
Ptilodactyidae	
Anchytarsus bicolor	xylophage
Diptera	
Simuliidae	
Simulium spp.	collector-filterer
Tipulidae	
Antocha sp.	collector-gatherer
Dicranota sp.	predator
Erioptera sp.	collector-gatherer
Hexatoma spp.	predator
Tipula sp.	shredder
Ephemeroptera	
Baetidae	
Baetis cf. brunneicolor	collector-gatherer
Ephemerellidae	
Drunella cf. cornutella	scraper
Ephemerella dorothea	collector-gatherer
Eurylophella spp.	collector-gatherer
Ephemeridae	
Ephemera guttulata	collector-gatherer
Heptageniidae	
Cinygmula subaequalis	scraper
Epeorus dispar	collector-gatherer
Heptagenia spp.	scraper
Stenacron spp.	scraper
Stenonema spp.	scraper
Leptophlebiidae	
Habrophlebia vibrans	collector-gatherer
Habrophlebiodes americana	collector-gatherer
Paraleptophlebia spp.	collector-gatherer
Siphlonuridae	
Ameletus lineatus	collector-gatherer

Table B2. (continued).

Taxa	Functional group
Isopoda	
Asellidae	
Lirceus fontinalis	shredder
Lepidoptera	
Ukn	shredder
Megaloptera	predator
Odonata	predator
Plecoptera	
Capniidae	
Allocapnia sp.	shredder
Paracapnia opis	shredder
Chloroperlidae	
Hastaperla brevis	collector-gatherer
Sweltsa/Suwallia spp.	predator
Leuctridae	
Leuctra truncata	shredder
Nemouridae	
Amphinemura spp.	shredder
Peltoperlidae	
Peltoperla cf. maria	shredder
Perlidae	
Acroneuria carolinensis	predator
Perlodidae	
Isoperla cf. holochlora	predator
Remenus bilobatus	predator
Yugus/Malirekus spp.	predator
Trichoptera	
Hydropsychidae	
Cheumatopsyche sp.	collector-filterer
Diplectrona modesta	collector-filterer
Hydropsyche sp.	collector-filterer
Lepidostomatidae	
Lepidostoma sp.	shredder
Limnephilidae	
Goera spp.	scraper
Neophylax spp.	scraper
Pycnopsyche sp.	shredder
Philopotamidae	
Wormaldia sp.	collector-filterer
Polycentropodidae	
Polycentropus sp.	predator
Rhyacophilidae	
Rhyacophila spp.	predator

APPENDIX C

Table C1. Regression coefficients (± 95% confidence intervals) of length-weight (dry) equations for a number of aquatic insects along with coefficient of determination (r^2), sample size, and reference.

Taxa	Ln α	β	R^2	N	Ref
Coleoptera					
Elmidae					
Stenelmis spp.	-5.66 (.)	3.01 (.)	0.99	4	ACB
Eubriidae					
Ectopria nervosa	-5.26 (0.716)	3.48 (0.499)	0.90	23	JRS
Psephenidae					
Psephenus herricki	-6.17 (0.622)	3.79 (0.330)	0.99	11	JRS
Diptera					
Simuliidae					
Simulium sp.	-5.34 (0.876)	2.55 (0.390)	0.85	26	LAS
S. spp.	-4.83 (.)	1.68 (.)	0.93	5	ACB
Tipulidae					
Tipula abdominalis	-5.30 (0.917)	2.36 (0.370)	0.93	19	LAS
Ephemeroptera					
Baetidae					
Baetis sp.	-5.51 (0.483)	3.02 (0.284)	0.96	20	JRS
B. sp.	-5.71 (0.501)	3.20 (0.810)	0.90	36	LAS
Ephemerellidae	-5.12 (0.142)	3.03 (0.100)	0.98	70	JRS
Ephemerella dorothea	-5.13 (0.394)	3.00 (0.288)	0.98	13	JRS
E. invaria	-5.00 (0.385)	2.96 (0.240)	0.98	13	JRS
E. rotunda	-4.94 (0.319)	2.97 (0.218)	0.99	9	JRS
Eurylophella temporalis	-5.12 (0.405)	2.76 (0.420)	0.91	18	LAS
Ephemeridae					
Hexagenia munda	-5.08 (0.726)	2.89 (0.340)	0.89	21	LAS
Heptageniidae	-4.83 (0.256)	2.91 (0.157)	0.93	102	JRS
Epeorus pleuralis	-4.34 (1.407)	2.61 (0.819)	0.85	10	JRS
E. rubidus/subpallidus	-4.89 (0.189)	2.81 (0.118)	0.99	16	JRS
E. sp.	-4.62 (0.410)	2.71 (0.254)	0.94	29	JRS
Heptagenia aphrodite	-4.02 (0.694)	2.33 (0.120)	0.85	90	LAS
Stenacron interpunctatum	-5.01 (0.292)	2.81 (0.260)	0.94	45	LAS
S. sp.	-4.70 (0.210)	2.86 (0.131)	0.98	42	JRS
Stenonema annexum	-4.85 (0.262)	2.87 (0.080)	0.94	128	LAS
S. modestum	-4.65 (0.663)	2.91 (0.385)	0.97	11	JRS
Leptophlebiidae	-5.96 (0.286)	3.18 (0.158)	0.98	38	JRS
Habrophlebiodes americana	-5.80 (0.561)	3.09 (0.387)	0.97	11	JRS
Paraleptophlebia moerens	-4.98 (1.257)	2.69 (0.614)	0.84	17	JRS
Oligoneuridae					
Isonychia sp.	-5.99 (0.953)	3.13 (0.453)	0.91	22	JRS
I. sp.	-5.78 (0.439)	3.17 (0.070)	0.97	107	LAS

Table C1. (continued).

Taxa	Ln α	β	R^2	N	Ref
Isopoda					
Asellidae					
Lirceus fontinalis	-4.76 (0.811)	2.84 (0.521)	0.85	24	JRS
Plecoptera					
Capniidae					
Allocapnia sp.	-5.28 (0.936)	2.84 (0.450)	0.89	16	LAS
Chloroperlidae					
Leuctridae					
Leuctra spp.	-4.76 (0.811)	2.84 (0.521)	0.85	24	JRS
Nemouridae					
Amphinemura delosa	-4.97 (0.461)	2.68 (0.318)	0.98	10	JRS
Peltoperlidae					
Peltoperla cf. maria	-2.88 (1.262)	2.08 (0.609)	0.78	17	JRS
Perlidae	-5.85 (0.305)	3.24 (0.143)	0.95	114	JRS
Acroneuria abnormis	-5.12 (0.426)	2.92 (0.180)	0.98	25	JRS
A. abnormis	-5.01 (0.803)	2.96 (0.940)	0.96	10	LAS
A. carolinensis	-5.67 (2.118)	3.22 (0.738)	0.97	6	JRS
A. spp.	-5.22 (0.304)	3.00 (0.125)	0.98	48	JRS
Eccoptura xanthenes	-6.03 (1.792)	3.44 (0.500)	0.97	7	LAS
Perlesta placida	-5.60 (1.532)	3.08 (0.450)	0.90	15	LAS
Phasgonophora capitata	-4.89 (1.417)	2.81 (0.689)	0.95	7	JRS
Perlodidae	-5.09 (0.363)	2.77 (0.181)	0.98	25	JRS
Isogenus sp.	-4.83 (0.693)	2.71 (0.430)	0.92	32	LAS
Isoperla bilineata	-5.03 (1.589)	2.69 (0.764)	0.90	9	JRS
I. clio	-5.45 (1.459)	2.90 (0.480)	0.87	19	LAS
I. spp.	-5.14 (.)	2.75 (.)	0.94	12	JRS
Taeniopterygidae					
Taeneopteryx burksi	-4.91 (0.903)	2.55 (0.800)	0.75	32	LAS
T. metequi	-5.12 (1.099)	2.85 (0.480)	0.95	12	LAS
Trichoptera					
Hydropsychidae					
Cheumatopsyche sp.	-6.16 (0.689)	3.07 (0.333)	0.97	15	JRS
C. sp.	-5.52 (0.694)	2.67 (0.900)	0.77	45	LAS
Diplectrona modesta	-5.81 (1.819)	2.78 (0.821)	0.79	15	JRS
Hydropsyche sp.	-6.07 (1.867)	3.11 (0.857)	0.79	17	JRS
H. sp.	-5.45 (0.766)	2.91 (0.650)	0.85	107	LAS
Macronema carolina	-5.81 (1.099)	3.09 (0.720)	0.91	8	LAS
Lepidostomatidae					
Lepidostoma sp.	-5.32 (0.490)	2.84 (0.290)	0.94	21	LAS
Limnephilidae					
Pycnopsyche guttifer	-5.04 (1.179)	2.74 (0.340)	0.93	16	LAS
Philopotamidae					
Chimarra atterima	-5.01 (0.803)	2.61 (0.610)	0.82	20	LAS

Table C1. (continued).

Taxa	Ln α	β	R^2	N	Ref
Polycentropodidae					
Phylocentropus spp.	-6.15 (.)	2.94 (.)	0.99	11	ACB
Polycentropus sp.	-5.63 (1.281)	3.00 (0.830)	0.79	15	LAS

JRS = present study
LAS = Smock (1980)
ACB = Benke et al. (1984)

Table C2. Calculation of Ectopria nervosa production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Crabapple Creek			
0-2	9.04	5.28	0.18				
				-10.19	0.05	-0.56	-3.35
2-3	19.23	14.37	2.88				
				-1.06	0.24	-0.26	-1.54
3-4	20.29	15.55	7.91				
				10.95	0.62	6.84	41.03
4-5	9.34	5.86	9.34				
				3.93	1.30	5.12	30.74
5-6	5.41	2.62	9.20				
				5.13	2.31	11.85	71.11
6-7	0.28	0.08	0.88				
				0.28	1.77	0.50	2.98
	total biomass=30.39				total production=140.98		
					variance=(294.39)		
				Bruce Creek			
0-2	2.83	1.55	0.08				
				1.76	0.06	0.11	0.66
2-3	1.07	0.25	0.14				
				-0.94	0.24	-0.23	-1.38
3-4	2.01	0.55	0.92				
				0.10	0.65	0.07	0.39
4-5	1.91	0.39	1.76				
				-1.06	1.32	-1.39	-8.36
5-6	2.97	0.81	5.58				
				1.78	2.35	4.19	25.15
6-7	1.19	0.28	3.51				
				1.19	1.72	2.04	12.26
	total biomass=12.00				total production=28.72		
					variance=(45.61)		

Table C3. Calculation of Elmidae spp. production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Crabapple Creek			
0-2	8.37	3.81	0.17	-9.50			
					0.03	-0.30	-1.80
2-3	17.87	14.11	0.89	11.02			
					0.09	0.95	5.73
3-4	6.85	2.27	1.03	3.73			
					0.22	0.82	4.90
4-5	3.12	1.12	1.00	1.72			
					0.45	0.78	4.67
5-6	1.40	0.31	0.90	0.55			
					0.75	0.42	2.49
6-7	0.85	0.17	0.76	0.85			
					0.94	0.80	4.81
		total biomass=$\overline{4.74}$				total production=$\overline{20.80}$	
						variance=(6.54)	
				Bruce Creek			
0-2	1.64	1.17	0.05	-1.08			
					0.04	-0.05	-0.32
2-3	2.72	0.82	0.16	2.09			
					0.10	0.20	1.43
3-4	0.63	0.14	0.10	0.00			
					0.24	0.00	0.00
4-5	0.63	0.14	0.22	0.63			
					0.47	0.30	2.09
5-6	0.00	0.00	0.00	-0.25			
					0.72	-0.18	-1.27
6-7	0.25	0.06	0.20	0.00			
					1.11	0.00	0.00
7-8	0.25	0.06	0.38	0.25			
					1.23	0.31	2.15
		total biomass=$\overline{1.12}$				total production=$\overline{4.08}$	
						variance=(1.23)	

Table C4. Calculation of Psephenus herricki production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Crabapple Creek			
0-2	8.28	9.08	0.08	2.69			
					0.03	0.08	0.68
2-3	5.59	2.03	0.45	-10.98			
					0.15	-1.64	-14.79
3-4	16.57	25.58	4.64	4.71			
					0.43	2.02	18.22
4-5	11.86	20.16	7.83	7.23			
					0.89	6.46	58.15
5-6	4.63	2.92	5.60	3.76			
					1.75	6.58	59.21
6-7	0.87	0.40	2.20	0.00			
					3.39	0.00	0.00
7-8	0.87	0.15	3.94	0.62			
					5.49	3.40	30.63
8-9	0.25	0.06	1.66	-0.25			
					8.28	-2.07	-18.62
9-10	0.50	0.25	5.15	0.50			
					3.21	1.60	14.44
			total biomass=31.55			total production=147.92	
						variance=(1388.30)	

Table C4. (Continued).

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
			Bruce Creek				
0-2	2.07	0.77	0.04	0.19			
					0.03	0.01	0.05
2-3	1.88	0.94	0.11	0.50			
					0.12	0.06	0.50
3-4	1.38	0.70	0.36	-0.18			
					0.44	-0.08	-0.63
4-5	1.56	0.55	1.15	1.06			
					1.09	1.15	9.23
5-6	0.50	0.13	0.80	0.09			
					1.83	0.16	1.32
6-7	0.41	0.10	0.86	-0.50			
					2.87	-1.44	-11.49
7-8	0.91	0.22	3.58	0.66			
					4.97	3.28	26.23
8-9	0.25	0.06	1.57	0.25			
					2.51	0.63	5.01
	total biomass=8.47				total production=30.22		
					variance=(116.42)		

Table C5. Calculation of Simuliidae spp. production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Crabapple Creek			
0-2	5.26	3.29	0.11	-14.66			
					0.03	-0.46	-3.25
2-3	19.92	18.49	1.00	-0.59			
					0.08	-0.05	-0.32
3-4	20.51	27.45	2.46	15.00			
					0.16	2.38	16.67
4-5	5.51	3.66	1.16	-1.37			
					0.28	-0.38	-2.67
5-6	6.88	7.74	2.55	3.05			
					0.45	1.38	9.63
6-7	3.83	2.59	2.11	3.58			
					0.69	2.48	17.33
7-8	0.25	0.06	0.22	0.25			
					0.93	0.23	1.63
		total biomass=9.59				total production=39.03	
						variance=(32.32)	
				Bruce Creek			
0-2	1.58	1.21	0.03	-3.11			
					0.03	-0.11	-0.75
2-3	4.69	8.58	0.28	-3.16			
					0.08	-0.26	-1.80
3-4	7.85	7.12	0.86	2.34			
					0.15	0.36	2.49
4-5	5.51	7.68	1.16	3.95			
					0.27	1.09	7.60
5-6	1.56	0.85	0.56	-3.09			
					0.45	-1.39	-9.71
6-7	4.65	5.41	2.60	4.37			
					0.63	2.77	19.42
7-8	0.28	0.08	0.20	0.28			
					0.85	0.24	1.66
		total biomass=5.70				total production=18.92	
						variance=(19.03)	

Table C6. Calculation of Hexatoma spp. production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Crabapple Creek			
0-5	40.12	44.91	3.21	4.24			
					0.22	0.92	5.53
5-10	35.88	15.68	21.17	27.44			
					1.02	28.04	168.25
10-15	8.44	2.56	14.94	5.20			
					2.67	13.89	83.33
15-20	3.24	0.49	13.06	2.99			
					6.14	18.36	110.18
20-25	0.25	0.06	2.34	-0.32			
					11.12	-3.56	-21.35
25-30	0.57	0.32	7.53	0.57			
					3.63	2.07	12.43
		total biomass=62.24				total production=358.37	
						variance=(1603.82)	
				Bruce Creek			
0-5	5.89	2.22	0.71	-2.34			
					0.27	-0.64	-3.83
5-10	8.23	5.66	5.10	4.51			
					1.09	4.89	29.37
10-15	3.72	0.89	7.07	3.21			
					3.18	10.20	61.18
15-20	0.51	0.26	2.71	0.39			
					6.84	2.67	16.00
20-25	0.12	0.02	1.06	-0.16			
					9.91	-1.59	-9.51
25-30	0.28	0.08	3.12	0.28			
					3.34	0.93	5.61
		total biomass=19.76				total production=98.82	
						variance=(537.37)	

Table C7. Calculation of Baetis brunneicolor production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Crabapple Creek			
0-2	15.62	7.44	0.31	-21.23			
					0.03	-0.74	-3.68
2-3	36.85	36.00	2.21	21.50			
					0.09	2.04	10.20
3-4	15.35	6.17	2.30	8.46			
					0.23	1.94	9.69
4-5	6.89	8.31	2.41	3.57			
					0.47	1.69	8.45
5-6	3.32	3.14	2.12	3.32			
					0.80	2.66	13.28
	total biomass=9.36				total production=37.94		
					variance=(27.03)		
				Bruce Creek			
0-2	5.23	1.58	0.10	-5.93			
					0.03	-0.19	-1.13
2-3	11.16	4.46	0.56	-0.69			
					0.09	-0.06	-0.37
3-4	11.85	6.10	1.90	3.43			
					0.23	0.80	4.80
4-5	8.42	3.32	2.86	6.07			
					0.47	2.83	16.99
5-6	2.35	0.82	1.50	2.26			
					0.85	1.92	11.53
6-7	0.09	0.01	0.10	0.09			
					1.06	0.10	0.57
	total biomass=7.03				total production=32.40		
					variance=(16.00)		

Table C8. Calculation of Cinygmula subaequalis production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Crabapple Creek			
0-2	42.62	62.24	1.70	13.57			
					0.07	0.90	5.40
2-3	29.05	48.68	3.20	18.42			
					0.18	3.29	19.74
3-4	10.63	13.35	3.08	7.57			
					0.40	3.00	17.97
4-5	3.06	1.38	1.65	0.53			
					0.74	0.39	2.36
5-6	2.53	2.30	2.58	1.27			
					1.25	1.58	9.49
6-7	1.26	0.57	1.92	1.26			
					1.23	1.55	9.32
	total biomass=14.13				total production=64.28		
					variance=(81.52)		
				Bruce Creek			
0-2	2.46	0.77	0.10	1.39			
					0.07	0.10	0.39
2-3	1.07	0.26	0.13	0.54			
					0.18	0.10	0.39
3-4	0.53	0.14	0.14	-0.04			
					0.40	-0.02	-0.06
4-5	0.57	0.32	0.34	0.57			
					0.77	0.44	1.75
	total biomass=0.71				total production=2.46		
					variance=(0.91)		

Table C9. Calculation of Epeorus dispar production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Crabapple Creek			
0-2	36.20	21.31	1.45	3.39			
					0.07	0.22	2.25
2-3	32.81	19.11	3.61	10.10			
					0.18	1.77	17.73
3-4	22.71	23.63	6.36	11.08			
					0.40	4.47	44.65
4-5	11.63	6.29	6.75	5.88			
					0.76	4.46	44.56
5-6	5.75	2.16	5.69	2.51			
					1.23	3.08	30.79
6-7	3.24	1.39	4.92	1.46			
					1.97	2.87	28.74
7-8	1.78	1.32	4.54	0.40			
					2.89	1.16	11.57
8-9	1.38	0.43	4.53	0.35			
					3.68	1.29	12.90
9-10	1.03	0.35	4.26	0.46			
					4.76	2.19	21.89
10-11	0.57	0.32	3.12	0.57			
					2.34	1.33	13.33
		total biomass=45.23				total production=228.40	
						variance=(724.96)	

Table C9. (Continued).

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
			Bruce Creek				
0-2	34.51	20.40	1.38	-6.63			
					0.07	-0.46	-5.51
2-3	41.14	31.59	4.94	15.99			
					0.19	3.03	36.41
3-4	25.15	17.43	7.54	8.11			
					0.42	3.41	40.94
4-5	17.04	12.85	10.05	9.54			
					0.76	7.25	87.05
5-6	7.50	2.99	7.35	2.16			
					1.24	2.68	32.15
6-7	5.34	1.98	8.38	1.45			
					1.89	2.74	32.92
7-8	3.89	1.47	8.87	1.20			
					2.77	3.33	39.92
8-9	2.69	1.28	9.07	-0.21			
					3.85	-0.81	-9.70
9-10	2.90	0.47	12.76	1.02			
					4.91	5.01	60.16
10-11	1.88	0.86	10.32	1.31			
					6.46	8.46	101.54
11-12	0.57	0.14	4.33	-0.25			
					8.67	-2.17	-26.02
12-13	0.82	0.20	8.12	0.82			
					3.15	2.58	30.96
		total biomass=93.12			total production=420.81		
					variance=(2377.12)		

Table C10. Calculation of Habrophlebiodes americana production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Crabapple Creek			
0-2	13.13	11.78	0.26	-85.63			
					0.03	-2.71	-13.54
2-3	98.76	352.28	4.94	35.75			
					0.08	2.88	14.41
3-4	63.01	175.19	8.19	53.99			
					0.19	10.48	52.41
4-5	9.02	8.57	2.62	8.77			
					0.40	3.47	17.35
5-6	0.25	0.06	0.13	0.25			
					0.73	0.18	0.92
	total biomass=16.14				total production=71.56		
					variance=(86.78)		
				Bruce Creek			
0-2	0.37	0.08	0.01	-4.44			
					0.03	-0.14	-0.56
2-3	4.81	4.35	0.24	-2.31			
					0.08	-0.18	-0.72
3-4	7.12	14.50	0.85	6.27			
					0.17	1.09	4.34
4-5	0.85	0.58	0.21	0.85			
					0.50	0.43	1.70
	total biomass=1.31				total production=4.77		
					variance=(3.27)		

Table C11. Calculation of Ameletus lineatus production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
			Crabapple Creek				
0-2	6.93	6.77	0.14	-15.59			
					0.03	-0.49	-5.42
2-3	22.52	11.49	1.13	10.35			
					0.09	0.93	10.18
3-4	12.17	8.53	1.95	0.78			
					0.25	0.19	2.12
4-5	11.39	5.58	4.33	-1.13			
					0.50	-0.57	-6.27
5-6	12.52	4.32	8.39	1.67			
					0.89	1.48	16.33
6-7	10.85	5.13	12.80	3.47			
					1.42	4.94	54.38
7-8	7.38	1.45	12.69	-2.77			
					2.11	-5.86	-64.44
8-9	10.15	4.50	26.39	4.79			
					3.08	14.76	162.32
9-10	5.36	1.44	19.56	4.58			
					4.06	18.60	204.63
10-11	0.78	0.19	3.53	0.53			
					5.60	2.97	32.68
11-12	0.25	0.06	1.74	0.25			
					2.64	0.66	7.25
		total biomass=92.64				total production=413.75	
						variance=(1209.18)	

Table C11. (Continued).

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Bruce Creek			
0-2	2.11	0.62	0.04	-5.65			
					0.03	-0.20	-2.54
2-3	7.76	2.16	0.47	4.48			
					0.10	0.44	5.71
3-4	3.28	1.02	0.52	-1.65			
					0.25	-0.41	-5.29
4-5	4.93	1.42	1.87	1.01			
					0.50	0.51	6.63
5-6	3.92	1.04	2.63	-0.62			
					0.89	-0.55	-7.20
6-7	4.54	1.38	5.40	-0.03			
					1.43	-0.04	-0.56
7-8	4.57	0.74	7.91	-0.66			
					2.14	-1.41	-18.37
8-9	5.23	1.33	13.86	-0.62			
					3.08	-1.91	-24.83
9-10	5.85	1.88	20.94	2.94			
					4.29	12.61	163.95
10-11	2.91	1.29	14.96	1.60			
					5.90	9.44	122.70
11-12	1.31	0.27	8.87	0.49			
					7.62	3.73	48.52
12-13	0.82	0.20	7.03	0.54			
					10.03	5.42	70.44
13-14	0.28	0.08	3.29	0.28			
					3.43	0.96	12.48
		total biomass=87.79			total production=371.64		
					variance=(2328.83)		

Table C12. Calculation of Lirceus fontinalis production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
			Crabapple Creek				
0-2	2.01	1.31	0.04	1.82			
					0.04	0.07	0.61
2-3	0.19	0.01	0.01	-0.59			
					0.16	-0.09	-0.83
3-4	0.78	0.33	0.27	-11.65			
					0.47	-5.51	-49.62
4-5	12.43	65.58	7.96	5.06			
					0.83	4.21	37.86
5-6	7.37	7.56	7.96	1.66			
					1.33	2.21	19.88
6-7	5.71	3.13	9.36	4.55			
					2.04	9.29	83.58
7-8	1.16	0.25	2.95	0.07			
					3.14	0.22	1.98
8-9	1.09	0.34	4.23	0.84			
					4.93	4.14	37.29
9-10	0.25	0.06	1.57	0.25			
					2.50	0.63	5.63
		total biomass=34.35				total production=136.38	
						variance=(1106.26)	

Table C12. (Continued).

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
Bruce Creek							
0-2	16.58	60.98	0.66	9.84			
					0.07	0.68	7.50
2-3	6.74	3.09	0.81	-75.29			
					0.20	-14.75	-162.3
3-4	82.03	210.18	26.25	-25.21			
					0.44	-11.14	-122.5
4-5	107.24	162.98	65.42	48.38			
					0.80	38.72	425.91
5-6	58.86	60.86	61.80	39.04			
					1.33	52.01	572.06
6-7	19.82	11.42	33.50	10.12			
					2.08	21.05	231.55
7-8	9.70	3.88	24.83	3.53			
					3.08	10.88	119.67
8-9	6.17	3.90	22.89	4.89			
					4.35	21.29	234.21
9-10	1.28	0.70	6.54	0.87			
					5.73	4.98	54.81
10-11	0.41	0.10	2.63	0.29			
					7.33	2.13	23.40
11-12	0.12	0.02	1.01	0.12			
					2.89	0.35	3.82
		total biomass=246.34				total production=1388.1	
						variance=(8442.44)	

Table C13. Calculation of Hastaperla brevis production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Crabapple Creek			
0-2	3.00	0.76	0.06	-8.58			
					0.04	-0.32	-2.57
2-3	11.58	5.89	0.81	-0.61			
					0.12	-0.07	-0.56
3-4	12.19	5.18	2.32	2.46			
					0.26	0.63	5.07
4-5	9.73	5.39	3.41	5.94			
					0.45	2.70	21.59
5-6	3.79	1.52	2.24	3.25			
					0.73	2.38	19.05
6-7	0.54	0.14	0.49	0.54			
					1.18	0.64	5.08
7-8	0.00	0.00	0.00	-0.12			
					1.96	-0.24	-1.88
8-9	0.12	0.02	0.30	0.12			
					1.59	0.19	1.53
	total biomass=9.62				total production=47.31		
					variance=(31.89)		
				Bruce Creek			
0-2	1.55	0.46	0.05	-5.13			
					0.05	-0.24	-1.41
2-3	6.68	3.14	0.47	-1.17			
					0.11	-0.13	-0.79
3-4	7.85	4.54	1.41	5.37			
					0.25	1.35	8.09
4-5	2.48	0.68	0.87	1.97			
					0.43	0.84	5.04
5-6	0.51	0.26	0.27	0.39			
					0.82	0.32	1.91
6-7	0.12	0.02	0.15	0.12			
					1.13	0.14	0.81
	total biomass=3.21				total production=13.65		
					variance=(5.92)		

Table C14. Calculation of Leuctra truncata production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Crabapple Creek			
0-2	3.18	1.34	0.10	1.31			
					0.05	0.06	0.42
2-3	1.87	0.95	0.13	-1.07			
					0.11	-0.12	-0.82
3-4	2.94	0.79	0.50	1.74			
					0.23	0.41	2.84
4-5	1.20	0.60	0.38	0.98			
					0.44	0.43	3.01
5-6	0.22	0.02	0.13	0.10			
					0.79	0.08	0.55
6-7	0.12	0.02	0.12	-0.13			
					1.17	-0.15	-1.06
7-8	0.25	0.06	0.33	0.25			
					1.15	0.29	2.01
			total biomass=1.70			total production=6.95	
						variance=(2.42)	

Table C14. (Continued).

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Bruce Creek			
0-2	3.47	1.03	0.07	-4.58			
					0.04	-0.17	-1.54
2-3	8.05	6.98	0.56	3.71			
					0.11	0.39	3.53
3-4	4.34	2.13	0.69	3.40			
					0.26	0.88	7.93
4-5	0.94	0.17	0.39	0.19			
					0.52	0.10	0.89
5-6	0.75	0.12	0.48	0.66			
					0.78	0.51	4.63
6-7	0.09	0.01	0.09	0.00			
					1.15	0.00	0.00
7-8	0.09	0.01	0.13	0.09			
					1.75	0.16	1.42
8-9	0.00	0.00	0.00	-0.25			
					2.79	-0.70	-6.28
9-10	0.25	0.06	0.88	0.25			
					1.88	0.47	4.22
		total biomass=3.29				total production=14.81	
						variance=(12.87)	

Table C15. Calculation of Amphinemura delosa production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
			Crabapple Creek				
0-2	21.94	13.81	0.44	5.77			
					0.04	0.22	0.86
2-3	16.17	6.21	1.13	12.58			
					0.12	1.53	6.10
3-4	3.59	1.63	0.75	3.15			
					0.27	0.85	3.42
4-5	0.44	0.10	0.15	0.44			
					0.59	0.26	1.04
	total biomass=2.48				total production=11.42		
					variance=(1.76)		
			Bruce Creek				
0-2	79.07	128.78	2.37	-6.36			
					0.05	-0.29	-1.46
2-3	85.43	121.16	5.98	69.36			
					0.11	7.79	38.93
3-4	16.07	17.62	2.89	11.71			
					0.25	2.98	14.90
4-5	4.36	1.94	1.57	4.01			
					0.45	1.82	9.08
5-6	0.35	0.07	0.20	0.35			
					0.75	0.26	1.32
	total biomass=13.01				total production=62.78		
					variance=(31.12)		

Table C16. Calculation of Amphinemura wui production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Crabapple Creek			
0-2	26.99	32.79	0.81	-0.42			
					0.05	-0.02	-0.10
2-3	27.41	60.91	1.92	20.38			
					0.12	2.35	11.75
3-4	7.03	6.97	1.34	3.03			
					0.27	0.80	4.02
4-5	4.00	2.13	1.48	3.65			
					0.44	1.62	8.08
5-6	0.35	0.07	0.19	0.35			
					0.73	0.25	1.27
	total biomass=5.73				total production=25.03		
					variance=(14.81)		
				Bruce Creek			
0-2	10.68	4.08	0.32	-3.31			
					0.05	-0.16	-0.81
2-3	13.99	6.41	1.12	8.95			
					0.12	1.10	5.52
3-4	5.04	2.48	0.96	4.28			
					0.27	1.15	5.75
4-5	0.76	0.28	0.29	0.67			
					0.53	0.36	1.79
5-6	0.09	0.01	0.07	0.09			
					0.87	0.08	0.39
	total biomass=2.75				total production=12.63		
					variance=(2.96)		

Table C17. Calculation of Peltoperla cf. maria production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Crabapple Creek			
0-2	18.93	20.19	2.84	15.01			
					0.21	3.18	31.84
2-3	3.92	1.80	1.18	3.92			
					0.42	1.64	16.35
3-4	0.00	0.00	0.00	-0.28			
					0.81	-0.23	-2.28
4-5	0.28	0.08	0.32	-0.25			
					1.52	-0.38	-3.80
5-6	0.53	0.14	1.08	0.53			
					2.47	1.31	13.08
6-7	0.00	0.00	0.00	0.00			
					3.32	0.00	0.00
7-8	0.00	0.00	0.00	-0.25			
					4.06	-1.01	-10.14
8-9	0.25	0.06	1.12	0.25			
					5.17	1.29	12.91
9-10	0.00	0.00	0.00	-0.25			
					6.89	-1.72	-17.23
10-11	0.25	0.06	1.99	0.25			
					2.82	0.71	7.05
		total biomass=8.52				total production=47.79	
						variance=(221.86)	

Table C17. (Continued).

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Bruce Creek			
0-2	21.13	20.28	3.17	17.31			
					0.21	3.67	36.72
2-3	3.82	2.18	1.15	3.73			
					0.44	1.63	16.34
3-4	0.09	0.01	0.06	-0.03			
					0.94	-0.03	-0.28
4-5	0.12	0.02	0.17	-0.75			
					1.66	-1.24	-12.44
5-6	0.87	0.19	1.72	0.62			
					2.44	1.51	15.11
6-7	0.25	0.06	0.75	-0.47			
					3.32	-1.56	-15.62
7-8	0.72	0.15	2.65	0.63			
					4.34	2.73	27.32
8-9	0.09	0.01	0.46	0.00			
					5.42	0.00	0.00
9-10	0.09	0.01	0.52	-0.10			
					6.78	-0.68	-6.78
10-11	0.19	0.03	1.52	0.19			
					2.83	0.54	5.37
	total biomass=$\overline{12.16}$				total production=$\overline{65.75}$ variance=(185.08)		

Table C18. Calculation of Diplectrona modesta production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Crabapple Creek			
0-3	0.25	0.06	0.01	-0.62			
					0.06	-0.04	-0.43
3-4	0.87	0.29	0.09	-4.87			
					0.14	-0.71	-7.76
4-5	5.74	2.60	1.21	1.20			
					0.27	0.32	3.53
5-6	4.54	1.78	1.54	2.30			
					0.43	0.99	10.84
6-7	2.24	0.88	1.21	-0.69			
					0.67	-0.46	-5.08
7-8	2.93	0.66	2.43	1.03			
					1.00	1.03	11.35
8-9	1.90	0.51	2.30	1.81			
					1.30	2.36	25.91
9-10	0.09	0.01	0.13	0.00			
					1.86	0.00	0.00
10-11	0.09	0.01	0.22	0.00			
					2.81	0.00	0.00
11-12	0.09	0.01	0.29	0.00			
					4.22	0.00	0.00
12-13	0.09	0.01	0.50	0.09			
					2.36	0.21	2.34
			total biomass=9.92			total production=40.70	
						variance=(39.17)	

Table C18. (Continued).

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
			Bruce Creek				
0-3	0.53	0.11	0.03	-1.95			
					0.08	-0.16	-1.90
3-4	2.48	1.07	0.27	-11.34			
					0.15	-1.72	-20.68
4-5	13.82	7.57	2.90	7.57			
					0.27	2.02	24.27
5-6	6.25	4.16	2.12	-0.38			
					0.44	-0.17	-1.99
6-7	6.63	2.52	3.71	1.55			
					0.68	1.05	12.60
7-8	5.08	2.42	4.17	1.20			
					0.97	1.16	13.92
8-9	3.88	1.49	4.42	1.32			
					1.37	1.80	21.66
9-10	2.56	0.53	4.20	0.62			
					1.77	1.10	13.17
10-11	1.94	0.50	3.71	1.47			
					2.27	3.33	39.98
11-12	0.47	0.08	1.26	0.10			
					3.08	0.31	3.69
12-13	0.37	0.14	1.30	-0.29			
					3.78	-1.09	-13.14
13-14	0.66	0.12	2.67	0.66			
					2.01	1.33	15.94
		total biomass=30.78				total production=107.53	
						variance=(218.44)	

Table C19. Calculation of Neophylax cf. mitchelli production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size x 8 (mg/m^2)
			Crabapple Creek				
0-2	2.84	2.45	0.09	-0.88			
					0.05	-0.04	-0.32
2-3	3.72	1.00	0.26	-7.56			
					0.13	-0.96	-7.67
3-4	11.28	43.23	2.59	9.75			
					0.29	2.84	22.75
4-5	1.53	0.50	0.57	-0.21			
					0.52	-0.11	-0.87
5-6	1.74	1.30	1.25	-0.26			
					0.86	-0.22	-1.79
6-7	2.00	0.36	2.06	0.93			
					1.39	1.29	10.35
7-8	1.07	0.26	2.01	0.28			
					2.05	0.57	4.59
8-9	0.79	0.19	1.76	0.79			
					1.49	1.18	9.44
	total biomass=10.59				total production=36.48		
					variance=(104.81)		
			Bruce Creek				
0-2	1.06	0.26	0.02	-0.26			
					0.04	-0.01	-0.08
2-3	1.32	0.64	0.09	0.32			
					0.12	0.04	0.31
3-4	1.00	0.37	0.21	-3.22			
					0.29	-0.93	-7.47
4-5	4.22	5.68	1.69	3.22			
					0.56	1.80	14.39
5-6	1.00	0.50	0.78	-0.59			
					0.90	-0.53	-4.23
6-7	1.59	0.87	1.64	1.06			
					1.26	1.33	10.65
7-8	0.53	0.14	0.81	0.28			
					1.94	0.54	4.35
8-9	0.25	0.06	0.62	0.25			
					1.57	0.39	3.14
	total biomass=5.86				total production=21.07		
					variance=(42.85)		

Table C20. Calculation of Neophylax spp. production (mg/m^2 dry) by the size-frequency method with variance estimates.

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
			Crabapple Creek				
0-2	5.28	1.54	0.11	-1.90			
					0.04	-0.08	-0.81
2-3	7.18	6.04	0.65	-9.95			
					0.14	-1.37	-13.68
3-4	17.13	19.78	3.60	4.10			
					0.29	1.20	12.03
4-5	13.03	17.37	5.34	2.75			
					0.52	1.43	14.31
5-6	10.28	4.34	6.78	6.56			
					0.85	5.59	55.89
6-7	3.72	2.39	4.09	0.95			
					1.34	1.27	12.72
7-8	2.77	0.77	4.52	2.03			
					1.90	3.86	38.62
8-9	0.74	0.31	1.64	-0.25			
					2.69	-0.67	-6.74
9-10	0.99	0.35	3.24	0.74			
					3.54	2.62	26.19
10-11	0.25	0.06	0.96	0.25			
					1.96	0.49	4.89
		total biomass=30.92				total production=143.43	
						variance=(331.49)	

Table C20. (Continued).

Size class (mm)	N (m^{-2})	v(N) (m^{-2})	B (mg/m^2)	Number lost (m^{-2})	Mean weight (mg)	Weight loss (mg/m^2)	x No. size classes (mg/m^2)
				Bruce Creek			
0-2	4.88	2.46	0.10	3.44			
					0.04	0.14	1.51
2-3	1.44	0.32	0.12	-1.79			
					0.13	-0.24	-2.61
3-4	3.23	1.54	0.71	-0.74			
					0.30	-0.22	-2.47
4-5	3.97	2.43	1.67	0.21			
					0.54	0.11	1.24
5-6	3.76	1.10	2.59	0.87			
					0.88	0.76	8.41
6-7	2.89	1.60	3.24	2.64			
					1.24	3.27	35.97
7-8	0.25	0.06	0.34	0.00			
					1.64	0.00	0.00
8-9	0.25	0.06	0.49	-0.12			
					2.52	-0.30	-3.32
9-10	0.37	0.08	1.20	0.37			
					3.56	1.32	14.48
10-11	0.00	0.00	0.00	-0.25			
					4.32	-1.08	-11.88
11-12	0.25	0.06	1.19	0.25			
					2.18	0.55	6.00
	total biomass=11.64				total production=47.34 variance=(113.37)		

VITA

Joseph R. Schiller was born in Philadelphia, Pennsylvania on September 18, 1955. He attended elementary schools in that city and in several other states. He attended secondary school in Philadelphia, Pennsylvania and was graduated from George Washington High School in June 1973. The following September he entered Colorado State University, and in June 1978 received a Bachelor of Science degree in Zoology. In September 1979 he entered the University of Tennessee, Knoxville as a candidate for the Master of Science in Zoology. This degree goal was subsequently changed to a Doctor of Philosophy in Zoology which was awarded in March 1986. The author is a member of the American Association for the Advancement of Science.